Tracking butterflies for effective conservation

Chris van Swaay

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Content

1. General introduction

C.A.M. van Swaay

Biodiversity and conservation

The world's biodiversity is overwhelming. The total number of species is estimated at around 8.7 million, of which only 1.2 million are already catalogued (Mora et al., 2011). Most of this biodiversity is concentrated in the tropics, but even a small country in the temperate climate zone of Western Europe, the Netherlands, still holds approximately 47800 species (Noordijk et al., 2010). And although this little part of the world is one of the best investigated in the world, every year is good for newly discovered species. A thorough investigation of a small nature reserve near Tilburg in the province of Noord Brabant revealed 50 new species for the Netherlands and one for science (Van Wielink, 2011).

Nature is dynamic and species have always come and gone. As the human population grew, the impact of mankind on biodiversity has also grown. The Dutch landscape changed from forest dominated before the Roman Age to agriculture dominated from the Middle Ages onwards. This has led to a huge shift in the accompanying species, including the butterfly fauna (WallisDeVries & Van Swaay, 2009). From the 1950s onwards another large shift happened turning the semi-natural grasslands that dominated the countryside into intensively used *Lolium perenne* monocultures with no suitable habitat for any butterfly-species. This has led to a fear for a biodiversity crisis leading to the extinction of many species. Although there are many signs that such a crisis is well on the way (Conrad et al., 2006; Thomas et al., 2008), it is hard to measure it and even harder to halt and reverse this trend.

In a few decades Dutch grasslands changed from species-rich semi-natural grasslands to* Lolium perenne *monocultures.

Where a complete description of the biodiversity at a national scale is almost impossible, we can try to monitor the changes in biodiversity using indicators. Many indicators have been proposed, but Pereira et al. (2013) give an overview of the Essential Biodiversity Variables (EBV's) that could form the basis of monitoring programs worldwide. One of them is ´*Abundances and distributions of species populations: counts or presence surveys for groups of species*´, e.g. those that are easy to monitor or of special importance for ecosystem services, over an extensive network of sites, complemented with incidental data. Butterfly Monitoring

Schemes provide such data making butterflies very useful indicator species for measuring changes in biodiversity (Van Swaay et al., 2008).

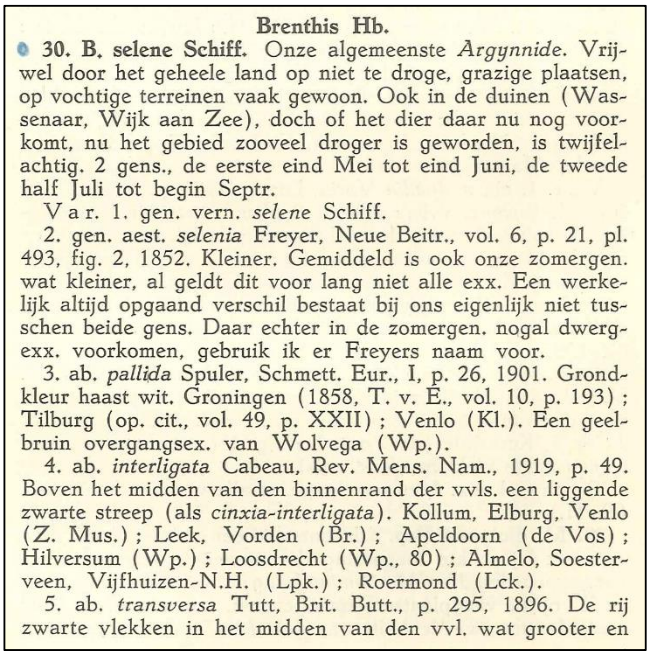
At the Convention on Biological Diversity (CBD) meeting in Nagoya (Japan, 18 to 29 October 2010) the Strategic Plan for Biodiversity 2011–2020 was adopted. It proposed five goals and 20 so-called Aichi targets. In line with this a new EU biodiversity strategy was adopted by the European Commission in May 2011. This provided a framework for the EU to meet its own biodiversity objectives and its global commitments as a party to the CBD. One of the main targets is to halt the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them as far as feasible, while stepping up the EU contribution to averting global biodiversity loss (EEA, 2012).
The strategy includes the development of a coherent framework for monitoring, assessing and reporting on progress in implementing actions. Such a framework is needed to link existing biodiversity data and knowledge systems with the strategy and to streamline EU and global monitoring, reporting and review obligations. Some indicators provide specific measurements and trends on genetic, species and ecosystem/landscape diversity, but many have a more indirect link to biodiversity. Very few were established specifically to assess biodiversity. The status indicators on species only cover birds and butterflies, since these are the only taxa/species groups for which harmonized European monitoring data are available (EEA, 2012)

It is important that one of the indicator species groups represents the insects. Insects are by far the most species-rich group of animals, representing over 50% of terrestrial biodiversity (Mora et al., 2011; Noordijk et al., 2010). Contrary to most other groups of insects, butterflies are well-documented, easy to recognize and popular with the general public. Thomas (2005) shows how well four different schemes, used successfully to assess changes in British butterflies (as well as Dutch butterflies), may be representative of other taxa. The four schemes include Red Lists, mapping schemes (atlases), Butterfly Monitoring Schemes and occasional surveys. Thomas (2005) also demonstrated that extinction rates in British butterflies are similar to those in a range of other insect groups over 100 years once recording bias is accounted for, although probably lower than in aquatic or parasitic taxa. It is concluded that butterflies represent adequate indicators of change for many terrestrial insect groups, but recommended that similar schemes be extended to other popular groups, especially dragonflies, bumblebees, hoverflies and ants. Comparisons with similarly measured changes in native bird and plant species suggest that butterflies have declined more rapidly than these other groups (Thomas et al. 2004).

Mountains can have a high butterfly diversity.

Butterfly research in the Netherlands

Butterflies have been popular in the Netherlands already for a long time. In the Golden Age (17th century) people like Johannes Goedaert, Stephen Blankaart and Maria Sybilla Merian studied butterflies and their metamorphosis (Bos et al., 2006). A major step forward was the publication of the first overview of Dutch butterflies by De Graaf (1853), soon followed by more studies (e.g. Snellen, 1867; Ter Haar, 1904).
Another landmark was set by the publication of the Catalogue of Dutch Macrolepidoptera that Lempke (1936) started and produced supplements until the late 1950s. Ten years later Frits Bink, from the Rijksinstituut voor Natuurbeheer (Research Institute for Nature Management) was one of the first scientists in Europe working in the field of nature conservation with a focus on butterflies.
With the start of the Landelijk Dagvlinder Project (Dutch National Butterfly Project) by Jan van der Made and Wim Geraedts in 1980 the way of observing and recording of butterflies changed drastically. Up to that moment a relatively small number of butterfly collectors had been active. From that year onwards a growing group of butterfly amateurs brought together a large amount of butterfly field records. In the six years of the Landelijk Dagvlinder Project the number of butterfly records (120,000) equalled that of the whole period up to 1980.
1983 marks the founding of De Vlinderstichting (Dutch Butterfly Conservation) and a following strong growth of the attention for butterfly conservation among the general public. Important other milestones were the publication of the first distribution atlas (Tax, 1989), a protection plan for Dutch butterflies (Ministerie van Landbouw, Natuurbeheer en Visserij, 1990), the re-introduction of two *Maculinea* species (Wynhoff, 2001) and the revised distribution atlas of butterflies in the Netherlands (Bos et al., 2006).



Brenthis Hb.

30. B. selene Schiff. Onze algemeenste *Argynnide*. Vrijwel door het geheele land op niet te droge, grazige plaatsen, op vochtige terreinen vaak gewoon. Ook in de duinen (Wassenaar, Wijk aan Zee), doch of het dier daar nu nog voorkomt, nu het gebied zooveel droger is geworden, is twijfelachtig. 2 gens., de eerste eind Mei tot eind Juni, de tweede half Juli tot begin Septr.

Var. 1. gen. vern. *selene* Schiff.

2. gen. aest. *selenia* Freyer, Neue Beitr., vol. 6, p. 21, pl. 493, fig. 2, 1852. Kleiner. Gemiddeld is ook onze zomergen. wat kleiner, al geldt dit voor lang niet alle exx. Een werkelijk altijd opgaand verschil bestaat bij ons eigenlijk niet tusschen beide gens. Daar echter in de zomergen. nogal dwergexx. voorkomen, gebruik ik er Freyers naam voor.

3. ab. *pallida* Spuler, Schmett. Eur., I, p. 26, 1901. Grondkleur haast wit. Groningen (1858, T. v. E., vol. 10, p. 193); Tilburg (op. cit., vol. 49, p. XXII); Venlo (Kl.). Een geelbruin overgangsex. van Wolvega (Wp.).

4. ab. *interligata* Cabeau, Rev. Mens. Nam., 1919, p. 49. Boven het midden van den binnenrand der vvls. een liggende zwarte streep (als *cinxia-interligata*). Kollum, Elburg, Venlo (Z. Mus.); Leek, Vorden (Br.); Apeldoorn (de Vos); Hilversum (Wp.); Loosdrecht (Wp., 80); Almelo, Soesterveen, Vijfhuizen-N.H. (Lpk.); Roermond (Lck.).

5. ab. *transversa* Tutt, Brit. Butt., p. 295, 1896. De rij zwarte vlekken in het midden van den vvl. wat grooter en

In the Catalogue of Dutch Macrolepidoptera (1936) Lempke gives a description for each species, with a focus on the variation. Here he mentions **Boloria selene** *to be common and widespread (see figure 2.1 for distribution maps of this species).*

Butterfly conservation essentials: what do we need to know?

In itself, butterfly conservation is a relatively new development. Especially birds have long generated numerous, popular and large organisations for the targeted conservation of this species group. For decades the conservation of butterflies was only treated from a general nature conservation perspective.
However in the 1970s it became more and more clear that the conservation of plants and birds was certainly not always beneficial for butterflies (Bink 1980). Butterfly conservation needs good quality data and a scientific approach to answer questions on planning and management, in what now is called evidence-based conservation (Pullin & Knight 2009; Thomas et al. 2011). Following a similar and earlier approach in the United Kingdom (Heath et al. 1984), the first step in the Netherlands was the Landelijk Dagvlinder Project, which delivered the first distribution atlas of butterflies in the Netherlands (Geraedts 1986; Tax 1989). It was the first attempt not only to get an overview of the distribution of butterflies, but also of their habitat use, ecology and conservation status. This book was soon followed by an overview of the ecological traits and habitat requirements of Northwestern European butterflies (Bink 1992).
From that moment the number of papers and books on butterfly conservation in Europe and other parts of the world has rocketed.

Effective species conservation is based on five pillars:

1. **Distribution**: where are they? Information on distribution, habitat preference and behaviour, including different life stages as well as migratory habits, is vital in order to organize conservation in an effective way. Only when we know where to find butterflies can we protect them or improve their habitat.
2. **Trend**: how are they doing? Species conservation is all about making choices: which species should be saved first. One of the essential parameters is the trend, both in population size as in distribution (see also in the IUCN Red List criteria (IUCN 2001) as well as in the reports for the reports on article 17 of the Habitats Directive (Evans and Arvela 2011)). For further analysis information on the trend in habitat quality and -availability can be necessary additional information.
3. **Drivers of change**: what are the causes? Knowing where the species can be found, how many species there are and what trend there is in their abundance, helps in focussing and may generate explanatory hypotheses, but does not, in itself, tell anything about the underlying causes. Scientific research on the ecology of species is then needed. Targeted indicators of ecologically relevant species groups can be effective and helpful tools to monitor changes, but the criteria for the choice of indicators also include items like policy relevance, public acceptance and affordability (Biała et al 2012). However, in many cases only detailed and long-term autecological research can reveal underlying mechanisms (e.g. Wynhoff, 2001).
4. **Conservation**: what can be done? Once the underlying causes for decline in species abundance have been identified, practical measures to overcome or mitigate the threats can be developed and tested. It is absolutely vital that these conservation measures are followed closely by long-term monitoring and regular evaluation. If necessary, they should be adapted according to the insights provided by monitoring data. Although there are many good examples of detailed research leading to successful changes in management (e.g. Brereton et al 2007; Thomas et al. 2009), this is not the case for the majority of declining butterflies, let alone for the multitude of other insect species whose distribution, population status and ecology remain poorly known. For an effective conservation of

biodiversity a sound scientific basis for evidence-based conservation should be extended to many more species.

5. **Communication**: how to raise awareness aiming at the general public? Only detailed reporting of the successes and failures of species conservation does really help us to learn from each other, not waste money and time and work toward an effective conservation. But there is more to it than writing a paper in a scientific journal. It is equally important to get this knowledge to the wardens and managers in the field in their own language as well as bringing it to the general public.

For the conservation of butterflies the five pillars are connected as the links of a chain or supporting a building: if we miss out on one of the pillars, the whole system might fail to achieve its conservation objective.
This thesis concentrates on pillars 1 and 2 in analysing the changes in distribution (Part I) and population trends obtained from butterfly monitoring (Part II) as well as pillar 3 via the development of indicators (chapter 7). Pillar 4 is discussed in more detail in Part III. Pillar 5 falls outside the scope of this thesis.

Below, these pillars will be briefly reviewed as an introduction to the following chapters.

Distribution

Although the number of people studying butterflies has always been much lower than for birds (compare the 150,089 members in 2013 for Vogelbescherming, the Dutch partner of Birdlife, with the 5,813 for De Vlinderstichting/Dutch Butterfly Conservation, Vroege Vogels Parade 2013, vroegevogels.vara.nl), there is still a remarkable amount of information available. The first lepidopterologists were mainly interested in extending their collection for taxonomic purposes. Already in the middle of the 19th century the first overviews were published, which also included the first attempts to give an overview of the distribution of all species as well as an indication of their rarity (De Graaf 1853). As time progressed, more and more of such data became available and in the 1930s Lempke (1936) could already give distribution lists consisting of Dutch butterfly communities.
However, not until the Dutch National Butterfly Project (Landelijk Dagvlinder Project) started in the early 1980s, accurate and up-to-date distribution maps, based on a 5 by 5 km grid, could be produced (Geraedts 1986; Tax 1989). For that project all known sources of old butterfly data were brought together and entered as records into a database. A total number of more than 120 000 pre-1980 butterfly records could be collected. This large dataset is not only the basis for our present knowledge on the potential distribution of Dutch butterflies, but also proved a valuable dataset for occupancy models.

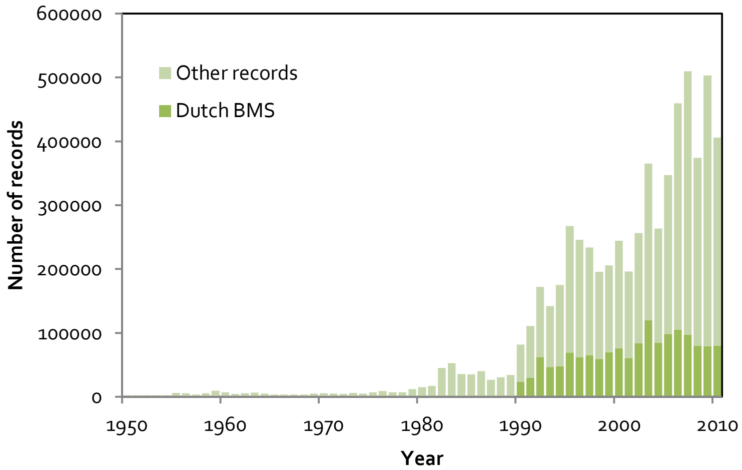


Figure 1.1: Number of records of butterflies per year in the Dutch National Database Flora and Fauna. Observations in the Dutch Butterfly Monitoring Scheme (Dutch BMS) are marked separately.

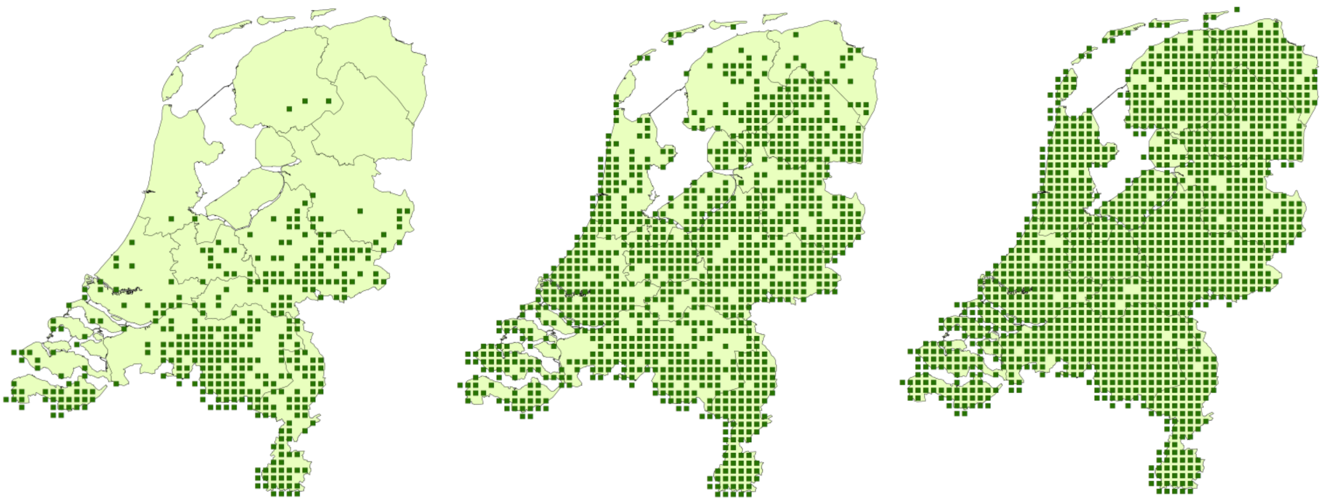


Figure 1.2: Distribution on a 5x5km scale of Polygonia c-album *in the Netherlands between 1981-1990 (left), 1991-2000 (middle) and 2001-2010 (right). Source: NDFF.*

Since the beginning of the 1990s the number of butterfly records has risen considerably, resulting in a new distribution atlas (Bos et al. 2006). And not only the number of records is high, more and more data became available on a 1 km grid scale and even more detail representing the real observations in the field in Landkaartje (www.vlindernet.nl/landkaartje), Telmee.nl and waarneming.nl. At present, Dutch butterfly distribution data are gathered in large numbers in these online portals. Also, the validation of these observations now follows standard procedures. Figure 1.1 shows the development of the number of records of butterflies per year in the National Database Flora and Fauna (NDFF).
In a well-investigated country like the Netherlands a distribution trend is not a change in the range or 'Extent of occurrence' as defined by IUCN (2008), but the change in the number of occupied squares (defined as the 'Area of occupancy' by IUCN, 2008). This can be both a range extension (figure 1.2 shows the range expansion of *Polygonia c-album* during its colonisation of the Netherlands) or filling up the gaps (figure 1.3 shows how *Pararge aegeria* expanded from its 'distribution islands' in the last twenty years to more or less cover the whole country).
However, size matters: large squares and long-time-periods reduce the sensitivity up to a point where a clear change is not detected anymore. But on the other hand: small squares and short time periods lead to a lot of missing values, as by far not every square in the Netherlands is visited a few times every year for a butterfly survey. Occupancy modelling (chapter 4) can be a way out of this dilemma.

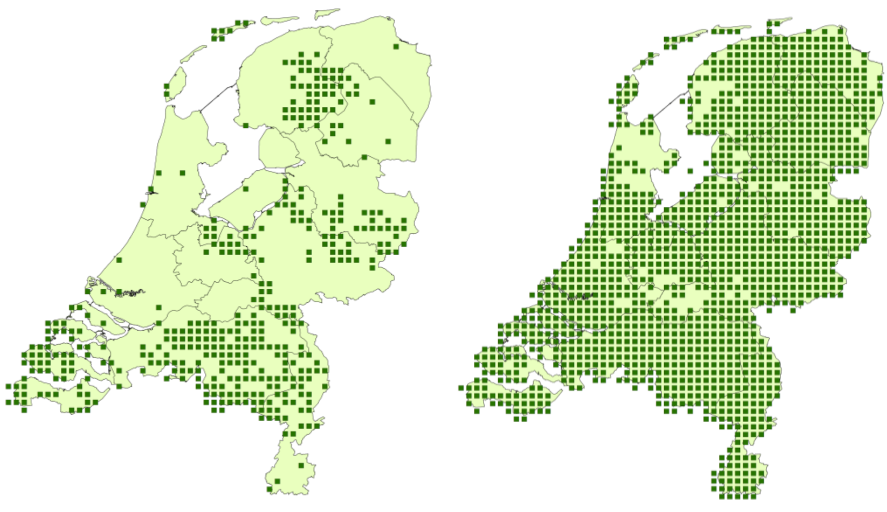


Figure 1.3: Distribution on a 5x5km scale of Pararge aegeria *in the Netherlands between 1980-1985 (left) and 2005-2010 (right). Source: NDFF.*

Distribution data on a national level have been collected for the purpose of the Red Data Book of European Butterflies (Van Swaay and Warren 1999) and the Red List (Van Swaay et al. 2010). This proved to be an effective and relatively easy and accurate way to collect information for the assessment of the Red List status. The network that was built for this purpose was also very helpful to produce the first overview of Prime Butterfly Areas in Europe (Van Swaay and Warren 1999). This also was one of the basic elements for the first overview on High Nature Value Farmland (Paracchini et al. 2008).

In most European countries the number of lepidopterologists, both professionals and volunteers, is lower than in the Netherlands (with the United Kingdom as the big exception). As a consequence, there are not many countries with distribution atlases comparable with the ones in the Netherlands, although in some regions a lot of data have been brought together. On a larger, European scale, two distribution atlases have been produced by Kudrna (2002; 2011). These have been very valuable in the assessment of the expected changes in butterfly distribution in the coming decades as a consequence of future climate change (Settele et al. 2008). In chapters 2 and 3 of this thesis a few methods to establish distribution trends with only limited data available are described and discussed.

Trend

By nature, populations are not stable (Thomas 1990; Traill et al. 2007). And this is certainly true for butterfly populations in a temperate climate, where all species have to go through some kind of hibernation and the weather can show large differences from year to year and even from day to day. After diapause all butterflies have to produce at least one generation of adults before the next winter – and some species produce even two or three, with the second and third generation generally larger than the previous one.

There are many hazards in the development from one generation to the other, besides environmental factors also predators, parasitoids and pathogens that will all influence the number of butterflies in the next generation. This necessitates a large reproductive potential. As a consequence of the combination of short generation times, large reproductive capacities and high variation in survival, the annual variability in butterfly population size is high compared to, for example, vertebrates..

Estimates of the real population size over time (e.g. the number of wild tigers in the world has dropped from 5000-7000 in 1998 to 3200 in 2009 as indicated by Chundawat et al. 2012), but this is already an unrealistic goal for vertebrates in most of the world, let alone for insects or even smaller and more abundant creatures. However, sampling for trends can offer a fairly easy way out, and the present Butterfly Monitoring Schemes in Europe all have followed this approach after the start of the first scheme in the United Kingdom in 1976 (Pollard & Yates, 1993). Figure 1.4 shows the population trends of *Polygonia c-album* and *Pararge aegeria* in the Netherlands as compared to the maps in figures 1.2 and 1.3. Sampling – in most cases on transects – provides no information on the exact population size, but can give reliable information on population trends. However, there are some possible sources of bias to overcome – most of these are discussed in chapters 5 and 6.

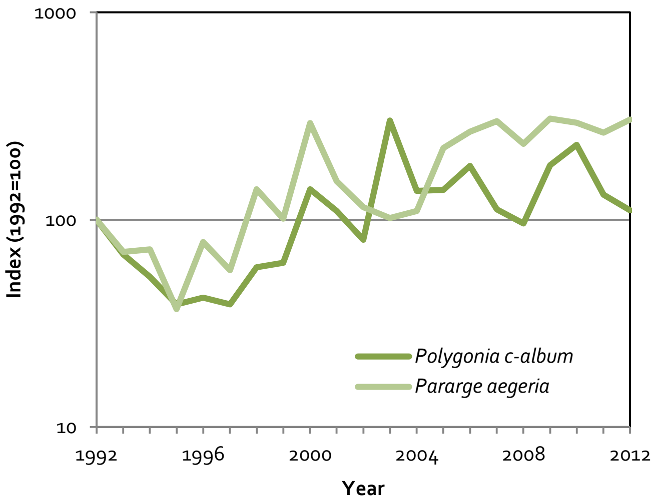


Figure 1.4: Population trend of Polygonia c-album and Pararge aegeria in the Netherlands. Source: De Vlinderstichting-Dutch Butterfly Conservation/CBS-Statistics Netherlands/NEM-Dutch Network for Ecological Monitoring.

The estimation of population trends based on transects requires a large number of such transects to obtain a reasonable statistical power. Van Strien et al. (1997) showed that, for many species, a minimum number of twenty transects is needed. Of course, when there are not enough populations to come up with twenty transects, it is best to try to count all sites. Another important phenomenon revealed by Van Strien et al. (1997) is that the variance of the first generation generally is much lower than in the second generation (although the numbers are usually lower in the first generation), making it important to focus on the first generations for an effective butterfly monitoring.

In many countries outside Northern and Western Europe, the number of volunteers needed for an effective Butterfly Monitoring Scheme is too low. This also applies to looking back in history. In the Netherlands the period between 1950 and 1980 regarding the number of volunteers and data, can be compared to the present situation in some Eastern and Southern European countries. If systematic data are available (e.g. from collections or literature) and have been brought together in a database, occupancy modelling (chapter 4) can offer a technique to establish reliable distribution trends per species. Nevertheless this method is also quite 'data-hungry', and that is when other methods can become more applicable (see chapters 2 and 3).

The second part of this thesis addresses pillars 2 (monitoring of abundance) and 3 (drivers of change). It focuses on the scientific basis of the monitoring population trends of butterflies in the Netherlands and Europe and using this information to build reliable indicators of changes in biodiversity.

Drivers of change

Butterflies live in a hazardous environment and are under a constant threat to get attacked by predators, parasitoids or pathogens, die of food shortage or poor quality food, cold or dry weather, and suffer from adverse effects of human land use by the application of pesticides, and direct mortality from mowing or grazing. As most butterflies only live as adults for a few weeks, most of this applies to one of the other, less mobile phases in the life of a butterfly: the egg, larvae or pupae. It is vital to find the bottlenecks in the life of a butterfly, as well as of populations as a whole, as this is the only way in which effective conservation measures can be developed. There are remarkable examples of successful conservation programmes, especially from the United Kingdom, and the long work of Jeremy Thomas on *Maculinea*'s (now *Phengaris*) and other butterflies should be mentioned as an example for many others (Thomas et al. 2009; 2011). In the Netherlands, this type of vital research has been taken up by Irma Wynhoff in the context of the reintroduction of *Phengaris teleius* and *P. nausithous* (Wynhoff, 2001; Wynhoff et al. 2011).

If we want to consider broader environmental threats like climate change or nitrogen deposition to a wider array of butterfly species, linking the results of trend analyses to other on-going changes can be a more effective way to move on. For example, WallisDeVries and Van Swaay (2006) showed that the combination of climate change and nitrogen deposition can explain the difference between the predominantly declining trends of early emerging spring butterflies (hibernating as adult or pupae) and the stable or increasing trends of later emerging species (hibernating as larvae or egg). In this approach, species are grouped according to species traits, with the potential to use the relevant species as indicators of a certain life history type. In a similar line, but from a habitat perspective, Brereton et al. (2009) (chapter 7) developed a butterfly indicator to record changes in grassland biodiversity.
Such an approach may also be followed in using species as indicators of environmental conditions. Thus, Oostermeijer and Van Swaay (1998, chapter 10) investigated a tool to study the underlying effects of changes in the species composition of butterfly communities in relation to environmental indicators of soil nitrogen, acidity and moisture. Devictor et al. (2012; chapter 9) took this indicator development one step further by integrating temperature indicator values for individual species into a community temperature index. They showed that it is possible to track the changes in composition of bird and butterfly communities as a function of climatic warming, thus providing an indicator to measure the synchronisation of these species groups to climate change.

Conservation

Although society wants to have evidence-based conservation relying on scientific research, the way to build this evidence is long and difficult and only seldom coincides with short-term project funding opportunities. The third part of this thesis provides first steps on which future scientists can build further.
Conservation is the translation from the former three pillars (distribution – trend – drivers of change) to real measures in the field. Typically, these are small-scale and often local changes in the management – a change in mowing or grazing regime, small scale sod cutting, etc. (e.g. WallisDeVries, 2004). On the other hand Butterfly Conservation Europe wants to make a difference in the large-scale changes with climate change and agricultural policy as the main items.
A Red List can bring together these items and show what species are most in need of conservation. Although efforts have been made to make the method to produce Red Lists more suitable for invertebrates, there are still species in need of urgent conservation which are missed by the IUCN rules (see chapter 12 for a discussion).

There are roughly two approaches for conservation:

- A species-based approach.
  In this approach the species is the point from which we start. For a long time this was the general way for nature conservation in large parts of the world. In Europe, the basis was often birds, in Africa large mammals and in the Arctic seas whales, so mostly large and attractive animals or plants like orchids. As butterflies are attractive as well, butterfly conservation organisations have managed to bring butterflies to the attention of nature conservation organisations more and more. Butterfly Conservation UK even went one step further and owns and manages its own nature reserves. In the Netherlands, butterfly-based conservation is an important way to preserve the last remaining population of some our most threatened butterflies (e.g. Wynhoff 2001; WallisDeVries 2004).
- A habitat-based approach.
  The last decade's habitat-based approaches for nature conservation have become more and more popular. And indeed, there is much in favour of preserving landscapes and habitats to ensure a firm basis for the survival of all characteristic species. However, the assumption that habitat management benefits all characteristic species is almost never tested. And with the economic incentive to minimize the costs of management, the preservation of rare species may be seriously jeopardized. Chapters 10 and 11 provide information for the development of habitat-based conservation of butterflies, in presenting both the habitat preferences for butterflies and a first approach to identify the most important areas in Europe to focus conservation policies on.

Should conservation of this calcareous grassland be based on its characteristic flora and fauna (here including species like Phengaris arion***) or solely on best habitat management for this type of grassland?***

Communication
Collecting data and researching butterfly ecology is vitally important for the conservation of butterflies, but it only remains paperwork without proper communication of the results back to the general public and to the people responsible for the management of butterfly habitats, from policy makers at European, national and local level down to farmers and nature wardens working in the field. Only with the emergence of butterfly conservation organisations in the UK and the Netherlands did this issue get the attention it deserved. Since 2004 Butterfly Conservation Europe and its partners try to canalise communication at the European level, by co-ordinating European research and conservation projects on butterflies, by communication on its website www.bc-europe.eu, facebook (www.facebook.com/ButterflyConservationEurope; see figure 1.5 for an example of the facebook page of De Vlinderstichting), Twitter (@europebutterfly) and its European Policy Advisor, Sue Collins, also together with other NGO's in the European Habitats Forum (www.eurosite.org/en-UK/content/european-habitats-forum).
Communication is important, but not a major focus of this thesis. Still it would be great if this thesis will also prove to be a valuable means of communicating the importance of butterfly conservation to conservation professionals as well as the general public.

Figure 1.5: Social media, like Facebook, can be an effective way to communicate with the public. This shows the Facebook page of De Vlinderstichting/Dutch Butterfly Conservation. Every message has the potential of reaching more than 7500 people directly (situation April 2014) and many more when a message is shared.

Outline of this thesis
This thesis consists of three parts. The first part (chapters 2-4) shows several methods to track changes in the distribution of butterflies. The second part (chapters 5-8) focusses on trends in butterfly abundance. The third part (chapters 9-12) shows how data gathered by volunteers and experts from all over Europe – and the Netherlands especially – can be used for the conservation of butterflies. The synthesis (chapter 13) will show that butterflies and the indicators developed with them, are excellent to follow the most important challenges for biodiversity in Europe in the next decades: climate change, agricultural intensification and abandonment.

Part I: Tracking changes in butterfly distribution

2. An assessment of the changes in butterfly abundance in the Netherlands during the 20th century

Slightly modified from: Van Swaay, C.A.M. (1990)
Biological Conservation **52,** 287-302

Abstract

Three methods of describing the changes in abundance of butterflies in The Netherlands are presented and discussed. The best proved to be the calculation of the percentage of the total number of investigated squares where the species was reported in a five-year period. Using this method six groups of species with a similar change in abundance are distinguished. Of the 63 species analysed, 29 (46%) have decreased or have become extinct, 17 (22%) have hardly changed their range and only 7 species (11%) seem to have expanded their range. Apart from this, 10 species (16%) fluctuate in range.

Until the middle of the 20th century Araschnia levana *was a rare and local butterfly. Nowadays it can be found all over the Netherlands. Figure 2.2 shows the expansion over the country.*

Introduction

Butterflies have been collected and studied by amateur and professional entomologists over many years. These historical data make butterflies an almost ideal group for studies on changes in the status of the different species. Recent investigations in The Netherlands showed that the distribution and abundance of many Dutch species of butterflies has decreased sharply (Geraedts, 1986). Of 71 native species, 15 have become extinct. A large proportion of the remainder is assumed to have declined. A serious problem in investigating the increase and decrease of species is the difference in the method and intensity used to collect the data. Up to the 1970s butterflies were only collected by a few entomologists, who were especially interested in rare species. Common butterflies were seldom reported. Many field observations are now made by a large group of people who have an interest in nature, and whose mobility and spare time are also much greater.

In this chapter different methods of describing changes in the abundance of butterflies are presented and discussed, followed by a survey of the abundance of species during this century by means of the most satisfactory method. After clustering, groups of species with a similar change of abundance are distinguished. Finally, predictions of future changes to be expected are made.

Material and methods

The basic material for this study consists of data from collections, literature and fieldwork, brought together for the Dutch Butterfly Mapping Scheme (Geraedts, 1986). At present almost 230 000 records of butterflies are available. This study uses only the data from the 71 native species, i.e. butterflies that have been present during the whole year in The Netherlands over a period of at least ten years.

The first problem in trying to quantify the abundance of a butterfly species is the enormous difference in the number of observations from year to year. There are several ways of tackling this problem:

1. Compare the numbers or range (e.g. the number of 5-km or 10-km squares) of a species before and after a certain date. This comparison is used in many atlas projects (e.g. Geraedts, 1986). Mostly two maps are presented, one with the old and one with the new distribution. Figure 2.1 shows this comparison for *Boloria selene*. This method does not include data on the length and intensity of investigations. In most cases a short, intensively investigated period is compared with a long, less intensively investigated period. For species that alternate periods of decline in distribution with ones of increase, the range as estimated in the longer period is overrated. The problem here is that the range is often taken from the peak distribution, even though this range may have been occupied for only a relatively short time.
2. Calculate for every year the running average of the number of butterflies or their distribution over a period of five or ten years. The running average over five years of a year n is the average of the value for the years $n-2$ to $n+2$.
3. Summarise the data in periods of five or ten years.

Method 3 is used for the data in this chapter. It is the easiest and includes the possibility of comparing different methods to estimate abundance in relation to time. The numbers are summarised over periods of five years. In this way the greatest precision can be reached. The relatively few data available for the beginning of this century did not allow a shorter period than five years.

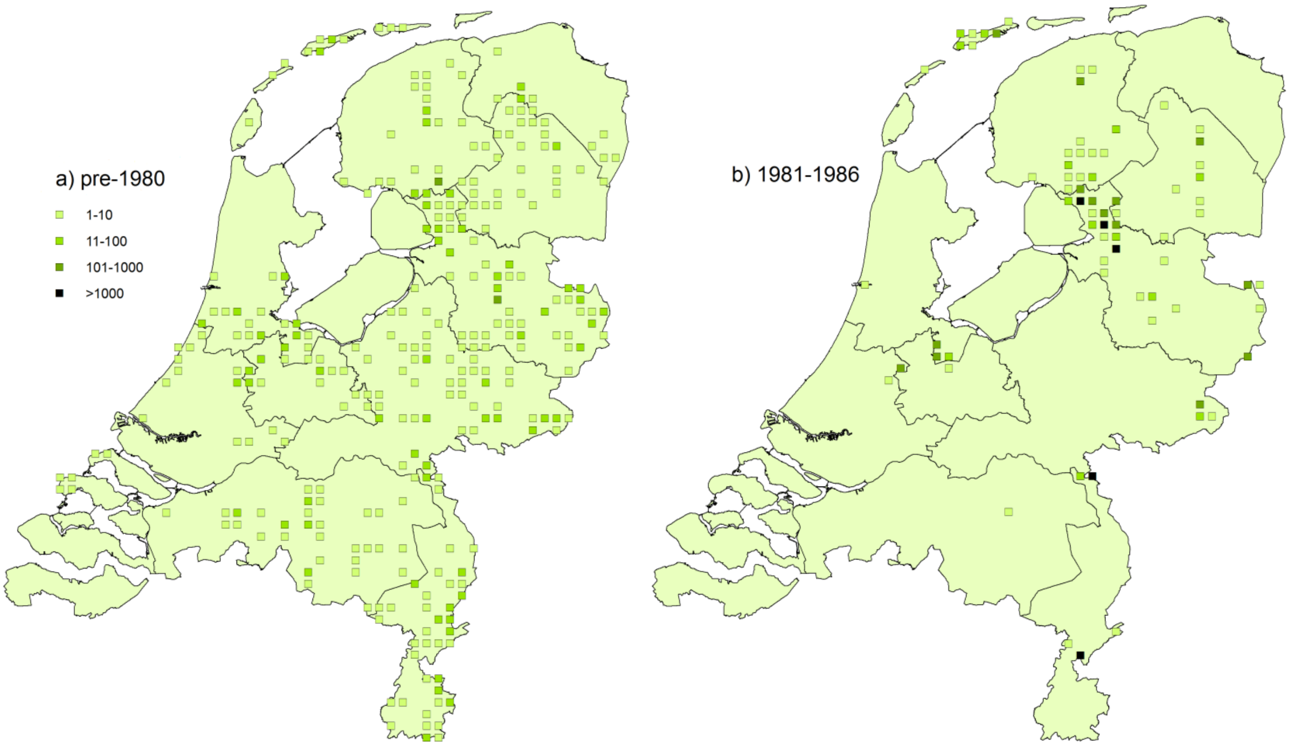


Figure 2.1: Distribution of* Boloria selene *in the Netherlands. a) Up to 1980 and b) 1981-1986.

The following methods can be used to estimate the variation of abundance of a butterfly species in time:

1. Calculate the total number of butterflies of a particular species observed over a five-year period. Disadvantage: No attention is paid to the changes in investigation intensity.
2. Calculate the percentage given by the numbers of a species recorded compared to the total number of butterflies observed in that period (see Meerman, 1987). It is based on the assumption that in every period each species is observed in relatively equal numbers. Disadvantages: Species which are temporarily relatively rare or abundant will give a distorted view. This can happen in two ways:
    a. For example, *Araschnia levana* was very rare until the beginning of the 1940s, when it expanded its range from the southeast. In the 1950s this butterfly could be seen in many places in The Netherlands and many observations were made. After a short decline *A.levana* became more and more common in the 1970s and can now be seen throughout the whole country. While this species is now regarded as common, it is relatively less reported by observers than it was in the 1950s.
    b. In earlier days, very common species were seldom caught, because they did not interest collectors. In more recent times naturalists have been encouraged to record all species, so that common species appear to have become relatively more numerous, which is not correct.
3. Calculate the percentage given by the number of squares in which a certain species was reported compared to the total number of investigated squares per period (see Turin & Den Boer, 1988). Advantages: (i) This method provides an opportunity to correct for investigation method and investigation intensity. (ii) Because this method is based on grid squares instead of numbers, the effect of a temporary

change in interest is less strong than in method 2. Disadvantages: (i) The period between 1981 and 1986 was intensively investigated for the Dutch Butterfly Mapping Scheme. This means that common species will be relatively overrated in this period. (ii) Rare, poorly dispersing species recorded from only a few localities and which were investigated very well in the past (e.g. *Boloria aquilonaris*) will score too highly during those periods which were less intensively investigated in the rest of the country. (iii) Since this method is based on grid squares, a decrease in the density of populations in the squares is not noticed. Therefore, the decrease of many species is noticed later than the actual start of that decrease.

In Figure 2.2 an example is given of these three methods using *Araschnia levana*. This shows that the peak in the 1950s for the recorded number of individuals (method 1, Figure 2.2a) and the percentage of the total number of recorded butterflies (method 2, Figure 2.2b) is earlier than the peak for the percentage of the total number of investigated squares (method 3, Figure 2.2c). This means that in the early years of the increase, when they attracted attention, many butterflies were recorded. Five years later the distribution in The Netherlands appeared larger, because observations were recorded for more squares, but there was less interest in catching or recording this species.

The last method combines the greatest accuracy with the fewest disadvantages and was chosen for this study. All data from native species between 1901 and 1980 are divided in five-year periods. After this the percentage of the total number of investigated squares where the species is reported in each period is divided into exponential classes (Table 2.1). This classification was developed by Geraedts (1986) as 'Square Frequency Class' (SFC) and was preferred to the classification of Van der Maarel (1971) and Westhoff & Weeda (1984), developed for botanical use. The fact that there are many more historical records for plants than for butterflies meant that their highest classes were seldom reached. The SFC is exponential, i.e. if a species sinks from class 7 to class 6, it is only observed in approximately half of the original number of squares. Eight native species could not be included in the analysis because of lack of data: *Heteropterus morpheus*, *Spialia sertorius*, *Satyrium pruni*, *Plebejus optilete*, *Phengaris alcon*, *Brenthis ino*, *Coenonympha hero* and *Hipparchia statilinus*. They are mainly species with a low dispersal rate, which were not observed for a long time and were suddenly 'rediscovered' on their old sites. These are considered to have been there all the time but not recorded.

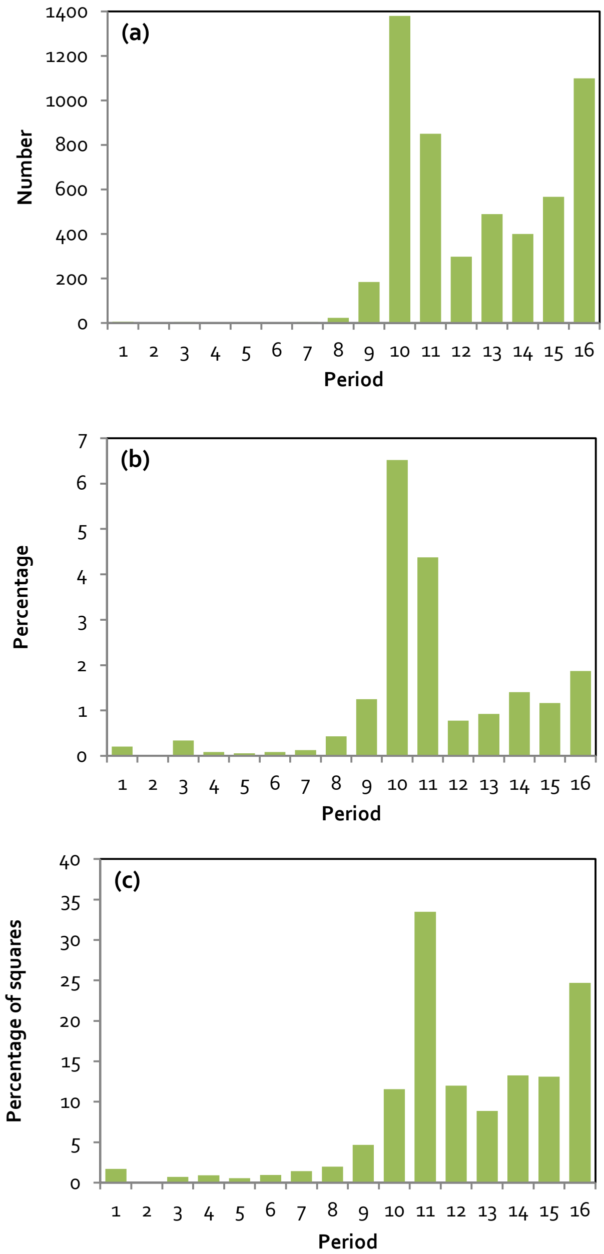


Figure 2.2: Abundance of Araschnia levana in five-year periods. Dates covered by the 16 periods in table 2. (a) Recorded number per period (method 1); (b) percentage of the total number of observed butterflies per period (method 2); percentage of the total number of investigated squares per period (method 3).

Table 2.1: Exponential Classification of the percentage of the total number of investigated squares where a species was reported (Square Frequency Class, SFC).

SFC	Upper limit (%)	Description
1	0.39	Extremely rare
2	0.78	Very rare
3	1.56	Rare
4	3.13	
5	6.25	
6	12.5	
7	25	Common
8	50	Very common
9	100	

Results

Dividing species into groups by means of their change in distribution and abundance in the 20th century is not easy since all species have their own history and ecological requirements. Six groups of species were identified with similar changes of abundance in time (Table 2.2). For each species the percentage of the total number of investigated squares in periods, where the species was reported between 1901 and 1980, is divided into SFC's. The measured SFC is only given for period 17 (1981-86).

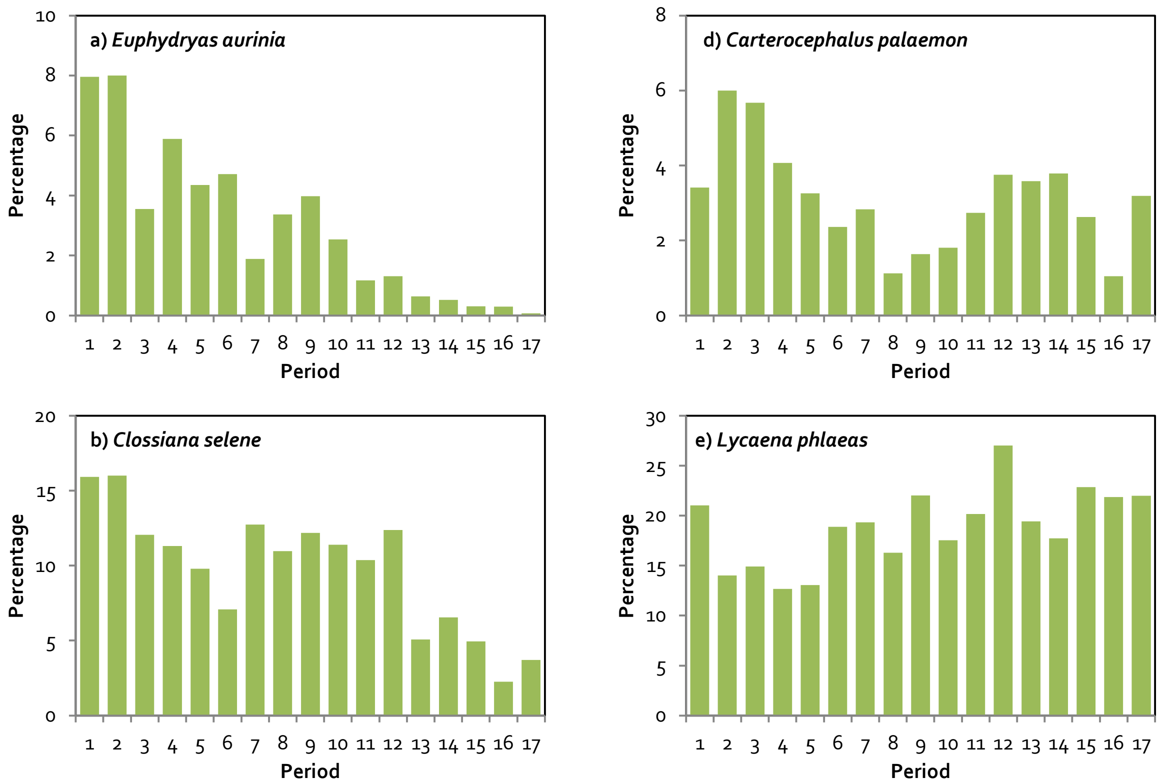
1. Species which have declined continuously since the beginning of the 20th century, most of which are now extinct or have only one or two local populations. Most of these species live on nutrient-poor grasslands. They all hibernate as larvae and have only one generation a year.
2. Species which have declined rapidly since the 1950s. Compared with this decade, only 10-50% of the squares are occupied today. Under normal circumstances most of these species have only one generation a year.
3. Species which have always been rare and declined slowly during the 20th century. Under normal circumstances they only have one generation per year in The Netherlands.
4. Species of which the distribution appears to fluctuate regularly. These species alternate periods of range expansion and reduction.
5. Species whose distribution has changed little or not at all. They are as common now as they were at the beginning of the 20th century.
6. Species which seem to have expanded their range. For some species this may be due to the fact that at the beginning of the 20th century they were seldom reported. They all hibernate as adult butterflies or as chrysalises. Except for *Gonepteryx rhamni* these species have two or more generations a year.

Figure 2.3 illustrates the first five groups with specific examples. Here the exact percentage of the total number of investigated squares is given for each period. Figure 2.2c provides an example for group 6.

Table 2.2: Changes in the percentage of the total number of investigated squares where a species was reported in the period 1901-1985. This percentage is divided into the Square Frequency Classes (SFC; table 2.1). This table was recalculated in 2011 and some figures can differ slightly from the original paper. Species names follow the 2011 version of the Fauna Europaea. Periods: 1=1901-1905, 2=1906-1910, etc.

	Five year period																
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**
Group 1																	
Lycaena hippothoe	5	4	2	4	5	2	4	2	3	1	1		1				
Phengaris arion	4	2	2		2	2	2	1	1	1			1				
Boloria euphrosyne	5	5	5	5	5	3	3	1	2	1	3	1		1			
Melitaea diamina	5	3	2	3	3	3	2	1	1	2	1	2					
Phengaris teleius	5	5	4	4	3	3	3	3	3	4	3	4	2	2			
Phengaris nausithous	4	4	3	4	4	3	3	2	3	4	4	3	2	2			
Thymelicus acteon	3	2	2	3	4	4	3	3	3	3	3	2	2	1	1	1	1
Euphydryas aurinia	6	6	5	5	5	5	4	5	5	4	3	3	2	2	1	1	1
Erynnis tages	6	6	5	5	5	5	4	4	4	5	5	4	3	2	2	1	1
Cyaniris semiargus	5	6	5	6	5	4	5	4	4	5	4	1	2	3	3	1	1
Melitaea cinxia	6	7	6	4	4	3	3	3	5	4	5	2	1	3	3	1	1
Group 2																	
Aporia crataegi	5	7	5	6	6	5	5	6	6	6	5	5	4	3	3	1	3
Nymphalis antiopa	6	6	5	5	6	6	5	6	6	5	5	5	3	3	3	4	3
Melitaea athalia	6	6	5	5	6	5	6	4	5	5	5	5	5	3	4	3	3
Argynnis paphia	5	4	4	4	5	5	5	4	5	5	4	4	5	4	3	4	3
Argynnis aglaja	6	6	5	4	6	6	5	5	6	5	5	5	4	4	4	2	4
Boloria selene	7	7	6	6	6	6	7	6	6	6	6	6	5	6	5	4	5
Pyrgus malvae	6	6	6	6	6	6	5	6	6	5	6	6	5	5	4	4	4
Coenonympha tullia	5	6	5	5	5	5	5	5	5	5	4	5	5	5	4	4	4
Argynnis niobe	5	5	3	5	5	5	5	5	5	5	5	6	4	5	4	4	4
Nymphalis polychloros	6	7	6	6	6	6	5	7	6	6	6	4	5	3	4	5	3
Satyrium ilicis	7	6	6	6	6	6	6	6	6	5	6	5	6	5	5	6	5
Lycaena tityrus	7	7	6	7	5	6	6	6	7	7	7	6	6	5	6	5	7
Group 3																	
Limenitis populi		2	4	2	2	4	4	2	2	1	1	1		1	1	1	1
Satyrium w-album	2	3		2	3	4	2	2	1	1	1	1	2		2	3	1
Cupido minimus	2	2			4	2	3		2	1	1		3	1		2	1
Plebejus idas	4	2	2	2		3	2	2	2	1	1	2	2	1	2	1	1
Coenonympha arcania	4	4	4	3			2	3	1	3	3	3	3	2	2	2	1
Boloria aquilonaris	4	3	2	3	2	2	3		3	1	1	3	2	3	2	3	2
Group 4																	
Apatura iris	5		3	3	3	4	4	4	4	4	3	3	3	3	2	4	4
Carterocephalus palaemon	5	5	5	5	5	4	4	3	4	4	4	5	5	5	4	3	5
Thecla betulae	5	6	4	5	5	4	5	5	4	4	5	3	4	4	4	3	4
Lycaena dispar			3	3	4	4	4	4	3	4	4	4	4	4	4	3	3
Aricia agestis	6	6	6	6	5	5	5	5	6	6	6	6	5	5	5	5	6
Hesperia comma	6	6	6	6	5	5	6	5	5	6	6	5	6	5	5	5	5
Limenitis camilla	6	5	4	4	5	6	6	6	6	5	6	5	5	5	5	5	6
Papilio machaon	6	6	7	7	5	5	7	7	7	7	6	5	5	6	6	6	5
Issoria lathonia	6	6	7	6	7	6	6	6	7	7	7	7	7	6	6	5	5
Polygonia c-album	6	4	3	4	5	6	6	6	6	7	7	6	6	6	6	6	7
Group 5																	
Plebejus argus	6	5	6	5	5	6	6	6	5	6	6	6	6	6	6	6	6
Callophrys rubi	6	6	6	6	5	6	6	6	6	6	6	6	6	6	6	6	7
Favonius quercus	6	6	5	6	6	6	6	6	5	5	6	6	6	6	6	6	7
Pyronia tithonus	6	6	6	6	6	5	6	6	6	6	6	7	6	6	6	6	8
Thymelicus sylvestris	6	6	6	6	6	6	6	6	6	6	6	6	6	5	6	6	7

Species	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17
	Five year period																
Thymelicus lineola	6	6	5	6	5	6	6	6	6	6	6	7	6	6	7	6	8
Pararge aegeria	5	5	6	6	5	5	6	6	6	7	7	7	6	6	7	7	8
Aphantopus hyperantus	6	6	5	6	5	6	6	6	6	6	6	7	7	6	6	6	8
Celastrina argiolus	6	7	6	7	6	7	7	7	6	6	7	7	6	7	6	7	8
Hipparchia semele	6	6	6	6	6	7	7	7	7	6	6	7	7	6	6	6	7
Anthocharis cardamines	7	7	6	6	6	6	7	6	6	6	6	6	6	7	7	7	8
Lasiommata megera	6	6	6	7	6	6	7	7	7	6	7	7	7	7	7	7	9
Ochlodes sylvanus	7	6	6	6	6	6	7	6	7	6	6	7	7	7	7	7	8
Maniola jurtina	7	7	6	6	7	7	7	7	7	7	7	7	7	7	7	7	9
Polyommatus icarus	7	7	7	7	7	7	7	7	7	7	7	7	7	7	7	7	9
Coenonympha pamphilus	7	7	7	7	7	7	7	7	7	7	7	8	7	7	8	7	9
Lycaena phlaeas	7	7	7	7	7	7	7	7	7	7	7	8	7	7	7	7	9
Group 6																	
Pieris brassicae	3	4	3	4	5	5	5	5	6	6	6	7	7	7	7	7	9
Pieris napi	4	5	3	5	5	5	5	6	6	6	6	7	7	7	7	7	9
Pieris rapae	4	3	3	5	6	5	5	5	6	6	7	7	7	7	8	7	9
Aglais io	4	5	5	5	2	4	5	6	5	6	6	7	6	7	8	8	9
Gonepteryx rhamni	6	5	6	6	6	7	7	6	6	7	6	7	7	7	7	8	9
Aglais urticae	2	4	5	5	5	5	5	6	6	6	6	7	7	7	8	8	9
Araschnia levana	4		2	3	2	3	3	4	5	6	8	6	6	7	7	7	8



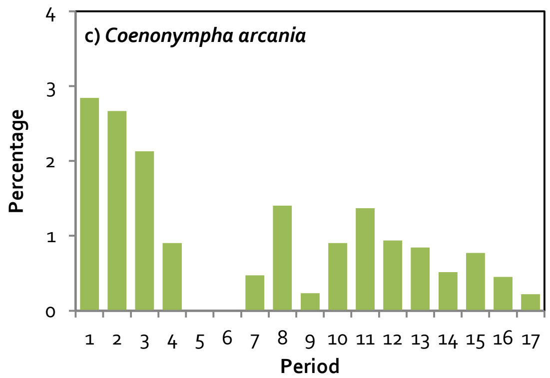


Figure 2.3: Percentage of the total number investigated squares per period for which the speces is reported. (a) Euphydryas aurinia *(group 1);* Clossiana selene *(group 2);* Coenonympha arcania *(group 3);* Carterocephalus palaemon *(group 4);* Lycaena phlaeas *(group 5). Figure 2.2c gives an example for group 6 (*Araschnia levana*).*

Discussion

Of the 63 species investigated, 29 (46%) have declined or have become extinct (groups 1, 2 and 3). Seventeen species (27%) hardly changed their range (group 5) and only seven species (11%) seem to have expanded their range (group 6). In addition, ten species (16%) fluctuate in range (group 4). For some of the last group, the peaks seem to become lower and the troughs deeper. These species might become endangered in the future.

In Table 2.3 the observed decline of the Dutch butterfly fauna is compared with plants and other groups of animals. This illustrates that butterflies and Orthoptera, as many other insects, are very vulnerable to changes in their environment because of their specialised lifecycles. All their requirements must be fulfilled every year, without fail, otherwise local populations will decline very rapidly, especially compared to birds. This stresses the value of insects such as butterflies as early warning indicators of environmental problems.

Table 2.3: The decline of butterflies in the Netherlands compared to plants and other groups of animals (Logemann, 1989).

Group	Number of species	Percentage of declining species
Bryophytes	535	36
Lichens	665	40
Vascular plants	±1450	34-54
Birds	155	26
Mammals	59	49
Butterflies	71	46
Orthoptera	39	49

For most of the species in group 1, which are now very rare or extinct, the decrease had already started at the beginning of the 20th century. As in the United Kingdom (Heath et al., 1984), loss of habitat is thought to be responsible for this major decline, which became stronger after the 1950s for most species. Figure 2.4 gives a short view on the changes in some semi-natural habitats in The Netherlands (CBS, 1976-78). During this century heathlands especially were cultivated, and today only about 20% is left compared to 1905. The total area of agricultural land in The Netherlands did not change very much, but at the beginning of this century almost all agricultural land maintained many butterfly species. Today the nutrient-poor, unimproved grasslands are restricted to small nature reserves, scattered all over the country. The rest of the arable land and pastures are unfit for butterflies. Isolation and poor management of the nature reserves have subsequently affected many of these isolated populations. Although isolated butterfly populations can survive very well for a long time (for example, *Maniola jurtina* on the Isles of Scilly (Dowdeswell et al., 1949)), a change in the management regime can lead to the lowering of the carrying capacity to a point where local extinction is very likely as a result of natural fluctuations caused by environmental changes. After this local extinction, isolation will reduce the chance of natural recolonisation. For example, in 1959 some 2000 specimens of *Euphydryas aurinia* were counted in its last population. Year after year the whole grassland was mown in the autumn and almost all the larval nests were removed. In 1975 only 50 specimens were left and in 1982 the last few were seen (Geraedts, 1986). It is clear that because the next population of this highly resident species is at least 300 km away, natural recolonisation is impossible.

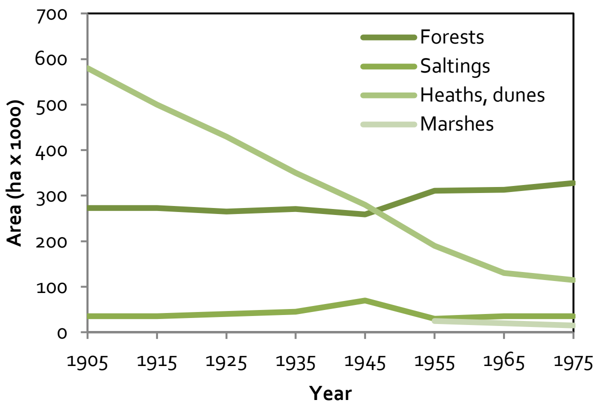


Figure 2.4: Changes in four semi-natural habitats the 20th century. Unimproved, semi-natural grasslands are not included, since they were always regarded as part of the agricultural grasslands.

The distribution of the species in group 2 changed little until the end of the 1950s. Since then the decrease has been substantial and only 10-50% of the original range is still occupied in the 1980 s. Habitat destruction outside nature reserves, and poor management inside, are the most important reasons for this sudden decline. Many of these species lived in a small-scale, not very intensively used agricultural landscape which has almost disappeared in The Netherlands since the spread of modern intensive agriculture. Also in England there was a sharp decline of many woodland and grassland species in the late 1950s and 1960s (Heath et al., 1984). Here, apart from habitat destruction, the disappearance of rabbits by myxomatosis was another important factor, stopping intense rabbit-grazing leading to higher vegetation and shrub invasion. In The Netherlands only the coastal dunes were temporarily influenced by myxomatosis.
The cause of the strong fluctuations of the species in group 4 is not precisely known and is presumed to be climatic. In the relatively warm period between 1930 and 1955, species like *Papílio machaon* and *Issoria lathonia*, which breed only in warm habitats, showed a clear peak in distribution. After this a period of decline began. In contrast, species such as *Carterocephalus palaemon* and *Lycaena dispar*, which favour relatively cool and moist breeding habitats, show a clear dip in the 1930s and 1940s and a peak in the wet 1960s.

It is striking that the species in group 1, which all show the strongest and longest decrease, hibernate as larvae and have only one generation per year. On the other hand none of the species in group 6 hibernate as larvae, but all as adult or as pupae. Bink & Siepel (1986) also established this fact. All species of group 6 except for *Gonepteryx rhamni*, have more than one generation per year. Without a change in agricultural use and management of nature reserves, it may be expected that all the remaining species of group 1 will soon become extinct. Their surviving populations are so small that any deterioration in the environmental situation will be fatal. Natural recolonisation is impossible when the nearest populations are too far away. Of the three species *Thymelicus acteon*, *Cyaniris semiargus* and *Melitaea cinxia*, not more than five adults were seen between 1980 and 1989. The last population of *Euphydryas aurinia* disappeared in 1982. Since then no butterflies of this species have been reported. To maintain these remaining species in The Netherlands biotope management will have to be adapted to the special demands of butterflies. The Protection Plan of the Ministry of Agriculture and Fisheries (Ministerie van Landbouw en Visserij, 1989) gives detailed information about this. For extinct species which are not able to colonise former sites where the management has been improved, reintroduction will be necessary. In the United Kingdom reintroductions have been carried out several times and proved to be successful under certain circumstances (Thomas, 1984). In 1990 *Phengaris teleius* and *P. nausithous* were reintroduced in a nature reserve in the south of The Netherlands, where *P. teleius* is still present but *P. nausithous* disappeared. Most species of group 2 are threatened by extinction in the near future. In less than 30 years their habitat has almost vanished from the Dutch countryside. Without any measures to protect and enlarge their habitats, the remaining populations will be prone to chance extinctions. This will lead to national declines as the chances of natural recolonísation are now much reduced due to the fragmentation and isolation of their habitats. How fast such a decline can occur is illustrated by the disappearance of *Carterocephalus palaemon* in England (Heath et al., 1984). By the time entomologists realised it was in danger, it had already disappeared from virtually all its sites. For these species a direct change of biotope management in favour of butterflies, as suggested in Ministerie van Landbouw en Visserij (1989), will be necessary. But more knowledge of the autecology of these species is also required. In addition, some of the fluctuating species in group 4 may become threatened, as the periods of decline became longer and deeper.

It is expected that the species in the groups 5 and 6 will be able to maintain themselves in The Netherlands, but their status must be watched carefully as a sudden fall in numbers may occur.

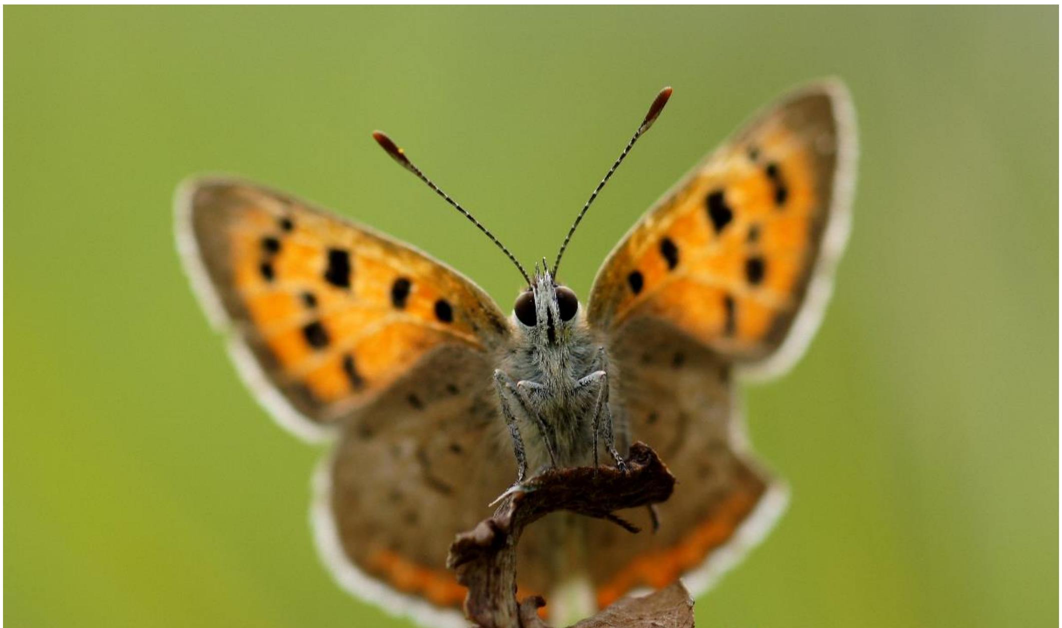
Acknowledgements
The author is very grateful to Mr J. G. van der Made, Ir J. Chr. Both and Drs I. Wynhoff for their stimulating interest and critical remarks. I also wish to thank Drs F. A. Bink, Dr P. J. den Boer, Mr H. Turin and Mrs R. Jurriëns-Macfarlane for the critical reading of the manuscript.

3. A new methodology for compiling national Red Lists applied to butterflies (Lepidoptera, Rhopalocera) in Flanders (N-Belgium) and the Netherlands

Slightly modified from: Maes, D. and Van Swaay, C.A.M. (1997)
Journal of Insect Conservation ***1****, 113-124*

Abstract

The compilation of the Red List of butterflies in Flanders and the Netherlands was based on two criteria: a trend criterion (degree of decline) and a rarity criterion (actual distribution area). However, due to the large difference in mapping intensity in the two compared periods, a straightforward comparison of the number of grid cells in which each species was recorded appeared inappropriate. To correct for mapping intensity, we used reference species that are homogeneously distributed over the country, that have always been fairly common and that did not fluctuate in abundance too much during the 20th century. For all resident species a relative presence in two compared periods was calculated, using the average number of grid cells in which these reference species were recorded as a correction factor. The use of a standardized method and well-defined quantitative criteria makes national Red Lists more objective and easier to re-evaluate in the future and facilitates the comparison of Red Lists among countries and among different organisms. The technique applied to correct for mapping intensity could be useful to other organisms when there is a large difference in mapping intensity between two periods.



Lycaena phlaeas*, the only reference species in both Flanders and in the Netherlands.*

Introduction

Since their conception in 1963 by Sir Peter Scott, Red Lists have been increasingly used as nature conservation tools (Collar, 1996). Red Lists or Red Data Books may have several uses: (i) to set up research programmes for conservation, (ii) to derive conservation priorities, and (iii) to propose protection for sites that are inhabited by threatened species (Mace, 1994; Collar, 1996). Their usage stresses that categorization of the different species should be based on reliable and objective criteria. In the past, almost all Red Lists were compiled on the basis of a best professional judgement by a group of experts. With their introduction for use in the compilation of international Red Lists by the International Union for the Conservation of Nature and Natural Resources (IUCN) (IUCN, 1994; Mace and Stuart, 1994), quantitative criteria have made their way into national Red Lists as well (e.g. Schnittler et al. (1994) in Germany). However, since many more data are available on vertebrates and on vascular plants, the proposed IUCN criteria are more easily applicable to these groups than to lower organisms, such as invertebrates or lower plants (Hallingbäck et al., 1995).
The method proposed by Strool and Depiereux (1989) for compiling the Red List of the Trichoptera in Belgium, which is based on the Chi^2-distribution, cannot be applied to the data set of the butterflies in Flanders and the Netherlands as, in order to use their method, the chance of finding a species should be equal in both compared periods: this condition is certainly not fulfilled since in the past more emphasis was on recording rare species while nowadays the common species represent the majority of the records. Recently, Avery at al. (1995) proposed another method for compiling the national Red List of British birds. The combination of three axes (axis 1 = the national threat status, axis 2 = the international importance and axis 3 = the European / global conservation status) was used as the basis for setting UK conservation priorities. However, due to lack of sufficient data, their method is difficult to use for invertebrates and in that case, they propose the use of qualitative information. Since the IUCN proposed a new approach for compiling Red Lists, it is recommended to develop methods that use quantitative criteria, even for invertebrates or other lower organisms.

In Flanders (N-Belgium) and the Netherlands, Maes et al. (1995) and Van Ommering (1994) recently proposed categories and criteria for the compilation of the respective national Red Lists. Although it is only a region of Belgium, we apply the terms 'country' and 'national' for Flanders for simplicity. The principal idea in this new method for compiling national Red Lists is that the present rarity of a species is compared with its rarity in a reference period. The distribution area in the reference period is considered as being the more or less natural distribution of most species. In the Netherlands, a lot of butterfly species showed a marked and strong decrease in the period 1950-1980 (van Swaay, 1990; chapter 1). In this period the Dutch landscape lost many suitable butterfly habitats due to the intensification of agriculture, acidification, etc. Therefore, the year 1950 marks the end of the reference period in the Netherlands. The start in 1901 was chosen arbitrarily. The number of butterfly records before this year was very low.
The method proposed for the compilation of the Red Lists in Flanders and the Netherlands uses a combination of the actual rarity and the degree of decline in distribution area to assign all resident species to a Red List category. The actual rarity is expressed as the extent of the present day distribution area and is measured as the number of grid cells in which a species was recorded in the period 1981-1995 in Flanders and the period 1986-1993 in the Netherlands (= period 2). The second criterion compares the present day distribution area with that in the period 1901-1980 in Flanders and 1901-1950 in the Netherlands (= period 1). Due to the large difference in mapping intensity between the two compared periods, we

had to design a way to compensate for this difference. In this paper we describe the general methodology for compiling the Red Lists in Flanders and the Netherlands. In particular, we introduce a technique that corrects for differences in mapping intensity among sampling periods. This technique may also be used to compare distribution areas of other groups of organisms when there is a large difference in mapping intensity between two sampling periods. The use of a standardized method with well-defined quantitative criteria, such as the one we propose in this paper, makes national Red Lists more objective and easier to re-evaluate in the future and facilitates the comparison of Red Lists among countries as well as among different groups of organisms.

Table 3.1: Red List categories and criteria used in Flanders and the Netherlands based on the IUCN criteria (IUCN, 1994); modified after Van Ommering (1994). Very rare species: presence <1% of grid cells; rare species: 1-5% of grid cells; fairly rare species: 5-12.5% of grid cells; common species: presence >12.5% of grid cells.

Red List category	Description
Extinct in the wild in Flanders/the Netherlands (EXF/EXN)	Species that did not have reproducing populations in Flanders/the Netherlands in the last ten years but have been recorded as such before. Some of these species are still observed as vagrants.
Critically endangered (CE)	Very rare species that decreased by at least 75% in distribution area between the two compared periods. In Flanders, species that have only a few isolated populations also qualify for this category.
Endangered (EN)	Very rare species that have decreased in distribution area by 50–75% between the two compared periods or rare species that have decreased by at least 50% in distribution area between the two compared periods.
Vulnerable (VU)	Very rare or rare species that have decreased in distribution area by 25–50% between the two compared periods or fairly rare species that have decreased in distribution area by at least 25% between the two compared periods.
Susceptible (SU)	Very rare species that have decreased in distribution area by less than 25% between the two compared periods (subcategory 'Rare' in Flanders) or common species that have decreased in distribution area by at least 50% between the two compared periods (subcategory 'Near-threatened' in Flanders).
Data deficient (DD)	Species for which there are insufficient data to place them in a Red List category.
Safe/Low risk (S/LR)	Rare and fairly rare species that have decreased in distribution area by less than 25% between the two compared periods or common species that have decreased in distribution area by less than 50% between the two compared periods.

Methods

The data for compiling the Red Lists of Flanders and the Netherlands were gathered by the Flemish Butterfly Study Group and by Dutch Butterfly Conservation respectively. At first, we gathered data from the literature and from museum and private collections. These data mainly date from before 1980 and comprise about 16 000 records in Flanders and about 125 000 in the Netherlands. Afterwards, both countries organized intensive campaigns with the help of numerous volunteers which greatly increased the data set. In Flanders, this butterfly mapping scheme started in 1991 and the complete data set now comprises about 145 000 records of 69 resident species. In the Netherlands, the mapping project started in 1981 and the complete data set now contains about 430 000 records of 70 resident species (Wynhoff and van Swaay, 1995). As the basis for mapping the distribution of each species, we used grid cells of 5x5 km both in Flanders (UTM projection, n=636) and the Netherlands (Amersfoort projection, n=1677).

Red list categories in Flanders and the Netherlands

The Red List categories in Flanders and the Netherlands are based on those of the IUCN (1994) and are given in Table 3.1. Both national Red Lists only refer to resident species, present in the country throughout the year and known to reproduce in the wild over a period of at least ten years. Thus, we excluded migratory species such as *Vanessa atalanta* (red admiral), *Cynthia cardui* (painted lady), *Colias hyale* (pale clouded yellow) and *Colias crocea* (clouded yellow). We used two criteria to classify species into the Red Lists of Flanders and the Netherlands: a rarity criterion and a trend criterion (Table 3.2).

Table 3.2: Classification scheme for the Red Lists of Flanders and the Netherlands; the number of grid cells that determine rarity are arbitrarily chosen.
Period 1: 1901-1980 in Flanders and 1901-1950 in the Netherlands.
Period 2: 1981-1995 in Flanders and 1986-1993 in the Netherlands.

Decline in distribution area between the two compared periods (%)	**Presence and percentage of grid cells**			
	<1%	1-5%	5-12.5%	>12.5%
	Number of grid cells Flanders			
	1-6	7-32	33-80	>80
	Number of grid cells the Netherlands			
	1-17	18-83	84-209	>209
76-100	Critically endangered	Endangered	Vulnerable	Susceptible
51-75	Endangered	Endangered	Vulnerable	Susceptible
26-50	Vulnerable	Vulnerable	Vulnerable	Safe/Low risk
≤ 25	Susceptible	Safe/Low risk	Safe/Low risk	Safe/Low risk

The rarity criterion is defined by the number of grid cells in which a species was recorded in period 2. The limits that determine rarity are arbitrarily chosen. For rare but fairly mobile species (e.g. *Aporia crataegi* (black-veined white), *Argynnis paphia* (silver-washed fritillary), *Issoria lathonia* (Queen of Spain fritillary), *Leptidea sinapis* (wood white), *Nymphalis polychloros* (large tortoiseshell) and *N. antiopa* (Camberwell beauty)), grid cells with single, vagrant individuals were excluded for compiling the Red Lists since they do not relate to populations.
The trend criterion is derived from the comparison between the actual rarity of a species and the extent of its distribution area in the past, expressed as the number of grid cells in which a species was recorded in period 1. However, due to the large

difference in mapping intensity between past and present, a simple comparison of the number of grid cells in the two periods is inappropriate. In Flanders there are about 13 000 records from the first period and about 130 000 from the second period, while in the Netherlands respectively 42 000 and 260 000 records are available. Furthermore, in the first period, mostly rare butterflies were collected or reported in literature, while after 1981 all species were recorded. To tackle the problem of the large difference in mapping intensity in the two compared periods, we use reference species to calculate a relative presence for each species in both periods. The decline in distribution area, calculated with the relative presences, will then be used as a trend criterion.

Determining reference species

For determining reference species, we used a method proposed by Latour and van Swaay (1992) that was already applied to determine the changes in butterfly abundances in the Netherlands (van Swaay, 1995).

First, for each resident species, the number of grid cells in which it was observed was counted per pentad (= period of five years; pentad 1 = 1901-1905, pentad 2=1906-1910, etc.). We subsequently expressed the number of grid cells in which a species was observed per pentad as a percentage of the total number of mapped grid cells in that pentad by

$$pp_{i,p} = 100 \times \frac{x_{i,p}}{n_p} \qquad (1)$$

where $pp_{i,p}$ is the presence in percentage of species i in pentad p, $x_{i,p}$ is the number of grid cells in which species i was recorded in pentad p and n_p is the total number of mapped grid cells (i.e. grid cells wherein at least one species was recorded) in pentad p. Secondly, we regressed the presence in percentage against pentad number for those species that are presently common, i.e. that were recorded in at least half of the total number of grid cells, and that are homogeneously distributed over the country. We applied this linear regression only for the periods before which the intensive mapping schemes started: up to and including pentad 18 (1986-1990) in Flanders and up to and including pentad 16 (1976-1980) in the Netherlands. Mapping intensity was considered more or less equal before the beginning of the intensive mapping schemes in both countries.

Reference species should then fulfill the following criteria: (i) the species should not have fluctuated too much during this century (i.e. the coefficient of determination $R^2 \geq 0.20$), (ii) the species should have been observed in at least 10% of the mapped grid cells at the beginning of this century (i.e. the intercept on the Yaxis $a \geq 10$), and (iii) the species should not have increased or decreased too strongly during the 20th century (i.e. $-1 <$ regression slope $b < 1$). The habitat in which the reference species occur is not taken into account.

Using reference species to compile the Red List

As a measure of the mapping intensity during the periods 1 and 2, the average number of grid cells in which the reference species were recorded in these two periods, was calculated as

$$\bar{r}_j = \frac{\sum_{t=1}^{n_r} x_{t,j}}{n_r} \qquad (2)$$

Where $\bar{r}_j$ is the average number of grid cells in which all reference species were recorded in period j, $x_{t,j}$ is the number of grid cells in which reference species t was recorded in period j and n_r is the total number of reference species. Using the average number of grid cells in which the reference species were recorded, we corrected for mapping intensity in both periods by calculating a relative presence for each species by

$$rp_{i,j} = 100 \times \frac{x_{i,j}}{\bar{r}_j} \qquad (3)$$

where $rp_{i,j}$ is the relative presence of species i in period j, $x_{i,j}$ is the number of grid cells in which species i was recorded in period j and $\bar{r}_j$ is the average number of grid cells in which the reference species were recorded in period j. By using the relative presences in both periods, the decline in distribution area for all resident species was estimated by

$$d_i = 100 - \left[100 \times \frac{rp_{i,2}}{rp_{i,1}} \right] \qquad (4)$$

where d_i is the decline in distribution area of species i, $rp_{i,1}$ is the relative presence of species i in period 1 and $rp_{i,2}$ is the relative presence of species i in period 2. Using the number of grid cells in which a species was recorded in period 2 ($x_{i,2}$) as a rarity criterion and the decline in distribution area (d_i) as a trend criterion, we classified all resident butterfly species into the Red List of Flanders and the Netherlands according to the scheme in table 3.2.

Table 3.3: Results of the linear regression on the species presence in percentage per pentad.
R^2=coefficient of determination, a=intercept on the y-axis, b=regression slope. When figures are in bold they fulfil the criterion for reference species.

	Flanders			the Netherlands		
	R^2	a	b	R^2	a	b
Aglais urticae	**0.56**	-1.1	2.13	**0.78**	-5.3	1.67
Araschnia levana	**0.67**	-7.6	2.02	**0.51**	-5.1	1.55
Celastrina argiolus	**0.22**	8.9	**0.71**	0.09	**11.8**	0.18
Coenonympha pamphilus	**0.61**	4.7	1.22	**0.57**	**11.9**	**0.71**
Gonepteryx rhamni	**0.48**	2.2	1.33	**0.75**	4.3	1.03
Inachis io	**0.60**	-2.4	2.06	**0.71**	-3.5	1.42
Lasiommata megera	**0.26**	**9.7**	**0.77**	**0.57**	6.29	**0.78**
Lycaena phlaeas	**0.30**	**12.1**	**0.86**	**0.29**	**14.9**	**0.39**
Maniola jurtina	**0.34**	8.3	**0.83**	**0.28**	**13.7**	**0.30**
Pararge aegeria	**0.42**	3.7	1.62	–	–	–
Pieris brassicae	**0.48**	1.6	1.43	**0.93**	-2.9	1.27
Pieris napi	**0.31**	**11.5**	1.26	**0.90**	-1.9	1.29
Pieris rapae	**0.43**	3.5	1.70	**0.89**	-3.7	1.51
Polygonia c-album	**0.56**	-2.5	1.51	–	–	–
Polyommatus icarus	**0.20**	**14.3**	**0.69**	0.05	**17.7**	**0.15**
Thymelicus lineola	**0.74**	-1.4	1.08	**0.43**	6.0	**0.35**

Results

The results of the linear regression analyses applied on the species presence in percentage per pentad are shown in Table 3.3. We determined three reference species in both countries: *Lasiommata megera* (wall brown), *Lycaena phlaeas* (small copper) and *Polyommatus icarus* (common blue) in Flanders and *Coenonympha pamphilus* (small heath), *L. phlaeas* (small copper) and *Maniola jurtina* (meadow brown) in the Netherlands.

With Equation (2), we calculated the average number of grid cells in which the reference species were recorded in the first and second period: in Flanders $\bar{r}_1$ is 154 and $\bar{r}_2$ is 379, and in the Netherlands $\bar{r}_1$ and $\bar{r}_2$ are 238 and 750 respectively. With equations (3) and (4) we subsequently calculated the relative presences and the declines in distribution area of all resident butterfly species (Appendix 1).

According to the scheme in Table 3.2, we then assigned all species to a Red List category (Appendix 3.1).

The use of these criteria results in 20 (29%) and 17 (24%) species in the 'Extinct' category and a further 25 (36%) and 30 (43%) species considered threatened (categories 'Critically endangered', 'Endangered', 'Vulnerable' and 'Susceptible') on the Red Lists in Flanders (Maes and Van Dyck, 1996) and the Netherlands (Wynhoff and van Swaay, 1995) respectively. In both countries, 23 species are presently considered as not threatened (Table 3.4).

Table 3.4: Number of species and percentage (in parentheses) per Red List category in Flanders and the Netherlands.

	Flanders		the Netherlands	
Extinct	20	(29)	17	(24)
Critically endangered	8	(12)	7	(10)
Endangered	6	(9)	11	(16)
Vulnerable	7	(10)	10	(14)
Susceptible	4	(6)	2	(3)
Data deficiënt	1	(1)	–	
Safe/Low risk	23	(33)	23	(33)

Discussion

The classification of the resident butterfly species in Flanders and the Netherlands into the national Red Lists, using the proposed method, has led to useful results for national nature conservation purposes. All butterflies listed as threatened on both Red Lists are indeed specialists of typical habitats that need urgent protection in Flanders and the Netherlands. The same classification method has already been successfully applied for compiling national Red Lists of a wide variety of other organisms like carabid beetles (Desender at al., 1995), amphibians and reptiles (Bauwens and Claus, 1996) and dragonflies (De Knijf and Anselin, 1996) in Flanders, and mammals (Hollander and van der Reest, 1994), birds (Osieck and Hustings, 1994) and grasshoppers (Ode, 1999) in the Netherlands.

Criteria like rarity and decline are used in most Red Lists, such as the British Red Data Books (Shirt, 1987; Bratton, 1991), but decline is usually described in a qualitative way ('rapid', 'continuous', etc.). In the newly proposed IUCN criteria (Mace and Stuart, 1994), the decline and the rarity criterion are used independently from one another; a species that has either declined in distribution area by at least 80% or that is very rare, is categorized as being 'Critically endangered'. Adopting the IUCN criteria for the national Red Lists of Flanders and the Netherlands would have placed respectively 14 and 15 species in the 'critically endangered' category, 7 and 12 species in the 'endangered' category and 1 and 6 species in the 'vulnerable' category. The additional criteria (the degree of potential immigration to counteract the decline) that the IUCN proposed for applying Red List categories at the national level (agreed at the National Red List Workshop in Gland, Switzerland, 23-24 March 1995) are difficult to apply to butterflies. Although some of the threatened or extinct butterflies are potentially fairly mobile, they do not seem to be able to found new populations in our countries. In Flanders and the Netherlands (but also in Germany (Schnittler at al., 1994)), the combined usage of the decline and rarity criteria, resulted in a classification into Red List categories on a national level that corresponded better with our judgements on butterfly threats in both countries than if IUCN criteria had been used.

Method for correcting for mapping intensity

Our method first identifies reference species which will consequently be used to calculate a decline in distribution area. Since reference species should be homogenously distributed over the country, it is not surprising that only grassland species qualify, since grasslands are the only habitats that are homogeneously distributed over both countries. Furthermore, these species are best represented in the family Lycaenidae and subfamily Satyrinae. The fact that the reference species are only found among grassland species strictly means that this method should only be used to evaluate the change in distribution area of grassland species. For species from other habitats, this method requires the additional assumption that butterflies in other habitats (e.g. forests, heathlands, etc.) were mapped with a similar effort as those in grasslands during both compared periods. In most European countries, 10 x 10 km grid cells are used for mapping invertebrates (e.g. Geijskens and van Tol, 1983; Desender, 1986; Emmet and Heath, 1989). The large amount of data in Flanders and the Netherlands made mapping possible on a 5 x 5 km scale. The imprecision of the older data (where often only the name of a town or an approximate location is given) did not allow the use of a finer scale. In Flanders, species that declined in distribution area on the basis of 5 x 5 km grid cells also did so when 10 x 10 km grid cells were used (r=0.951, n=67, $p<0.001$). The use of 5 x 5 km grid cells, instead of the usual 10 x 10 km grid cells, certainly allowed a better estimation of the decline in distribution area, but for most species we still underestimated the decline, since declines on distribution maps are only detected when all populations have disappeared from a grid cell (Thomas and Abery, 1995). The use of 10 x 10 km grid cells in Flanders instead of the 5 x 5 km grid cells, would have underestimated the decline of the rare species for 4% on average and for 36% on average for the intermediately rare species (see Thomas and Abery, 1995).
The method applied here to correct for mapping intensity, yielded informative results for the butterflies in Flanders and the Netherlands and proved to be useful for other groups of organisms that have been relatively well recorded throughout this century. This technique allowed a fairly good estimation of the decline in distribution area of rare and intermediately rare species, but not for the very common species. This is due to the fact that the latter were largely underrecorded in the past. Since we were compiling a list of threatened species, used to set conservation priorities in Flanders and the Netherlands, the presently common species were of a lesser concern for this purpose. For species with a very localized distribution area within both countries and which were recorded very well in the past, this method calculated a large decline in distribution area by correcting for mapping intensity (e.g. a decline of 73% and 59% for *Cupido minimus* and *Heteropterus morpheus* respectively in Flanders or 75% and 68% for *Boloria aquilonaris* and *Vacciniina optilete* respectively in the Netherlands). Most of these species inhabit typical and very localized habitats (chalk grasslands, peat bogs, etc.) and data suggest that their distribution area did not undergo changes.

Species in such cases are classified in the subcategory 'rare' of the Red List category 'susceptible' in Flanders because of their restricted distribution area in both the past and present.

Comparing the Red lists of Flanders and the Netherlands

The method we used to compile our Red Lists is repeatable and fairly objective. Furthermore, by using the same classification technique in Flanders and the Netherlands, their respective Red Lists become more easily comparable. However, the category 'Susceptible' has to be interpreted differently in the two countries. The four species in this category in Flanders have always had a restricted and localized distribution and are therefore put in the subcategory 'Rare'. The two species in the category 'Susceptible' in the Netherlands on the other hand, are still common but have decreased in distribution area by at least 50%. A second difference between both Red Lists is that the reference periods are not identical (1901-1980 vs. 1981-1995 in Flanders and 1901-1950 vs. 1986-1993 in the Netherlands). However, this does not affect the composition of the Red Lists: by applying the reference periods from the Netherlands to the data of Flanders, we obtained exactly the same Red List for Flanders as with the presently used periods. Since national Red Lists are used for shaping national public policy (Bean, 1987), each country can set different but appropriate reference periods.

Comparing the Red Lists of Flanders and the Netherlands shows that the group of threatened species is almost identical in both countries. Only two species were categorized differently: *Callophrys rubi* is categorized as 'Vulnerable' in Flanders but 'Safe/Low Risk' in the Netherlands, while *Papilio machaon* is 'Susceptible' in the Netherlands but 'Safe/ Low Risk' in Flanders. For the species that both countries have in common, the degree of decline is very similar (decline in distribution area in Flanders vs. the Netherlands, $r = 0.809$, $n = 63$, $p<0.001$). This fact is not surprising since both countries have a similar landscape and have undergone similar declines in the number of suitable butterfly habitats (heathlands, forest, nutrient-poor unimproved grasslands) through changes in agricultural management and building activities. Fragmentation of suitable habitats can strongly decrease or even stop the exchange of individuals between populations leading to a higher risk of extinction (e.g. Thomas and lones, 1993). Furthermore, a lot of butterfly habitats have deteriorated qualitatively through bad management or lack of management. A management plan for threatened butterflies, both on the population and on the landscape level, has already been produced in the Netherlands (Ministerie voor Landbouw, Visserij en Natuurbeheer, 1990) and is being prepared for Flanders (Maes and van Dyck, 2001).

A comparison of our Red Lists of butterflies with those in other Northwestern European countries or regions (not compiled with the new IUCN criteria) reveals that the group of extinct and threatened species varies from 51% (91 species) in Germany (Pretscher et al., 1984), over 63% (80 species) in Baden-Württemberg (Ebert, 1991) to 66% (51 species) in Wallonia, South-Belgium (Goffart et al., 1992). In Great Britain only 18% (10 species) of the species are extinct or threatened (Shirt, 1987). Although the global figures are alike (except for Great Britain) the proportion of extinct species is clearly higher in Flanders (29%) and in the Netherlands (24%) than in the other countries or regions. With 16 extinct species (16%), Wallonia (Southern Belgium) is intermediate between our countries and the other European countries or regions; Germany with only two (1%), Baden-Wiirttemberg with only four (3%) and Great Britain with only three extinct species (5%) do much better on this point. A comparison of threatened butterflies between countries is difficult due to different techniques used for compiling the national Red Lists. It would therefore be interesting to apply our technique to existing data sets in other countries or regions. Only by using the same technique will national Red Lists become comparable. Since a European Red List is being prepared, an objective and repeatable method, like the one proposed here, would be recommended.

Future Red Lists

Since butterfly distribution and threats are variable, Red Lists will have to be updated regularly (e.g. every ten years). Thanks to the large number of records that are gathered annually by numerous volunteers, the distribution of butterflies in Flanders and the Netherlands can now be easily monitored. The next Red Lists in both countries could, for example, compare the distribution of the species in the period 1991-2000 with that in the period 2001-2010. Due to the similar collecting technique (direct observations) and probably fairly similar mapping intensities, the number of grid cells of each species in both periods will be more easily comparable. Harmonization of the change-over date in future Red Lists should be aimed for throughout Europe and the year 2000 could be ideal for this purpose.
In the future, the Butterfly Monitoring Schemes in Flanders and the Netherlands, based on transect counts (Pollard and Yates, 1993) might be used in addition to the method proposed in this article, in order to take the trends in the numbers of individuals in the monitored populations of threatened butterfly species into account (van Swaay et al., 1997).

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Appendix 3.1

Number of grid cells in which the species was recorded in the periods 1901–1980 in Flanders and 1901–1950 in the Netherlands (x_1) and 1981–1995 in Flanders and 1986–1993 in the Netherlands (x_2) and their relative presence in both periods (rp_1, 100%=154 in Flanders and 238 in the Netherlands; rp_2, 100%=379 in Flanders and 750 in the Netherlands), the decline in distribution area (d, in percentage points) and the Red List category (RLC).

–=the species is not indigenous;

[v]all observations concern vagrant individuals;

[(x)]the number of grid cells with reproducing populations is given in brackets, the major part of the observations concern vagrant individuals;

[i]re-introduced species.

For the abbreviations of the Red List categories refer to Table 3.1.

	Flanders						Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	RLC	x_1	x_2	rp_1	rp_2	d	RLC
Aglais urticae	149	542	96.8	143.0	−48	S/LR	101	1008	42.4	134.4	−217	S/LR
Anthocharis cardamines	111	381	72.1	100.5	−40	S/LR	161	518	67.7	69.1	−2	S/LR
Apatura ilia	0	1	0	0.3	–	CE	–	–	–	–	–	–
Apatura iris	14	12	9.1	3.2	65	EN	31	28	13.0	3.7	71	EN
Aphantopus hyperantus	92	239	59.7	63.1	−6	S/LR	149	428	62.6	57.1	9	S/LR
Aporia crataegi	30	19[v]	19.5	5.0	74	EXF	98	16[v]	41.2	2.1	95	EXN
Araschnia levana	101	434	65.6	114.5	−75	S/LR	73	694	30.7	92.5	−202	S/LR
Argynnis paphia	30	21[(1)]	19.5	5.5	72	CE	59	28[v]	24.8	3.7	85	EXN
Aricia agestis	35	59	22.7	15.6	32	VU	107	149	45.0	19.9	56	VU
Boloria aquilonaris	–	–	–	–	–	–	9	7	3.8	0.9	75	CE
Brenthis ino	–	–	–	–	–	–	5	0	2.1	0	100	EXN
Callophrys rubi	53	56	34.4	14.8	57	VU	115	212	48.3	28.3	42	S/LR
Carcharodus alceae	14	0	9.1	0	100	EXF	–	–	–	–	–	–
Carterocephalus palaemon	38	64	24.7	16.9	32	VU	44	65	18.5	8.7	53	EN
Celastrina argiolus	115	366	74.7	96.6	−29	S/LR	166	707	69.8	94.3	−35	S/LR
Clossiana euphrosyne	13	0	8.4	0	100	EXF	31	0	13.0	0	100	EXN
Clossiana selene	51	1	33.1	0.3	99	CE	175	53	73.5	7.1	90	EN
Coenonympha arcania	3	0	2.0	0	100	EXF	14	2	5.9	0.3	95	CE
Coenonympha hero	4	0	2.6	0	100	EXF	4	0	1.7	0	100	EXN
Coenonympha pamphilus	156	328	101.3	86.5	15	S/LR	245	742	102.9	98.9	4	S/LR
Coenonympha tullia	16	5	10.4	1.3	87	CE	73	18	30.7	2.4	92	EN
Cupido minimus	6	4	3.9	1.1	73	SU	8	0	3.4	0	100	EXN
Cyaniris semiargus	64	2[(1)]	41.6	0.5	99	CE	57	1[v]	24.0	0.1	99	EXN
Erynnis tages	29	2[v]	18.8	0.5	97	EXF	64	2	26.9	0.3	99	CE
Eurodryas aurinia	20	0	13.0	0	100	EXF	64	0	26.9	0	100	EXN
Fabriciana adippe	9	0	5.8	0	100	EXF	–	–	–	–	–	–
Fabriciana niobe	7	0	4.6	0	100	EXF	76	41	31.9	5.5	83	EN
Gonepteryx rhamni	129	444	83.8	117.2	−40	S/LR	174	892	73.1	118.9	−63	S/LR
Heodes tityrus	91	4[v]	59.1	1.1	98	EXF	191	146	80.3	19.5	76	VU
Hesperia comma	29	22	18.8	5.8	69	EN	101	98	42.4	13.1	69	VU
Heteropterus morpheus	5	5	3.3	1.3	59	SU	6	14	2.5	1.9	26	VU
Hipparchia semele	82	79	53.3	20.8	61	VU	179	270	75.2	36.0	52	SU
Hipparchia statilinus	5	0	3.3	0	100	EXF	10	16	4.2	2.1	49	VU
Inachis io	144	543	93.5	143.3	−53	S/LR	87	1003	36.6	133.7	−266	S/LR
Issoria lathonia	69	25[(2)]	44.8	6.6	85	EXF	199	90	83.6	12.0	86	VU
Ladoga camilla	50	55	32.5	14.5	55	VU	104	95	43.7	12.7	71	VU
Lasiommata megera	146	347	94.8	91.6	3	S/LR	188	825	79.0	110.0	−39	S/LR
Leptidea sinapis	12	8[(1)]	7.8	2.1	73	CE	–	–	–	–	–	–
Limenitis populi	8	0	5.2	0	100	EXF	9	3	3.8	0.4	89	CE
Lycaeides idas	4	0	2.6	0	100	EXF	14	0	5.9	0	100	EXN

	Flanders						Netherlands					
Species	x_1	x_2	rp_1	rp_2	d	RLC	x_1	x_2	rp_1	rp_2	d	RLC
Lycaena dispar	–	–	–	–	–	–	15	6	6.3	0.8	87	CE
Lycaena phlaeas	150	388	97.4	102.4	−5	S/LR	237	742	99.6	98.9	1	S/LR
Maculinea alcon	25	23	16.2	6.1	63	EN	58	89	24.4	11.9	51	VU
Maculinea arion	–	–	–	–	–	–	9	0	3.8	0	100	EXN
Maculinea nausithous	–	–	–	–	–	–	14	2^{i}	5.9	0.3	95	EXN1
Maculinea teleius	9	0	5.8	0	100	EXF	17	2^{i}	7.1	0.3	96	EXN1
Maniola jurtina	133	414	86.4	109.2	−27	S/LR	233	765	97.9	102.0	−4	S/LR
Melanargia galathea	7	$18^{(1)}$	4.6	4.8	−5	SU	–	–	–	–	–	–
Melitaea cinxia	37	$6^{(4)}$	24.0	1.6	93	CE	63	1	26.5	0.1	99	CE
Melitaea diamina	6	0	3.9	0	100	EXF	18	0	7.6	0	100	EXN
Mellicta athalia	21	0	13.6	0	100	EXF	84	20	35.3	2.7	92	EN
Mesoacidalia aglaja	25	6^{v}	16.2	1.6	90	EXF	97	27	40.8	3.6	91	EN
Normannia ilicis	53	40	34.4	10.6	69	VU	115	96	48.3	12.8	74	VU
Nymphalis antiopa	34	18^{v}	22.1	4.8	79	EXF	94	15^{v}	39.5	2.0	95	EXN
Nymphalis polychloros	65	$40^{(10?)}$	42.2	10.6	75	EN	139	30	58.4	4.0	93	EN
Ochlodes venatus	122	312	79.2	82.3	−4	S/LR	174	503	73.1	67.1	8	S/LR
Palaeochrysophanus hippothoe	0	1	0	0.3	–	CE	22	0	9.2	0	100	EXN
Papilio machaon	126	310	81.8	81.8	0	S/LR	204	248	85.7	33.1	61	SU
Pararge aegeria	134	493	87.0	130.1	−50	S/LR	135	513	56.7	68.4	−21	S/LR
Pieris brassicae	138	493	89.6	130.1	−45	S/LR	88	873	37.0	116.4	−215	S/LR
Pieris napi	165	525	107.1	138.5	−29	S/LR	102	965	42.9	128.7	−200	S/LR
Pieris rapae	153	558	99.4	147.2	−48	S/LR	81	1011	34.0	134.8	−296	S/LR
Plebejus argus	63	40	40.9	10.6	74	VU	111	191	46.6	25.5	45	VU
Polygonia c-album	110	439	71.4	115.8	−62	S/LR	141	576	59.2	76.8	−30	S/LR
Polyommatus icarus	167	402	108.4	106.1	2	S/LR	267	651	112.2	86.8	23	S/LR
Pyrgus armoricanus	3	0	2.0	0	100	EXF	–	–	–	–	–	–
Pyrgus malvae	42	11	27.3	2.9	89	EN	132	38	55.5	5.1	91	EN
Pyronia tithonus	99	358	64.3	94.5	−47	S/LR	146	451	61.3	60.1	2	S/LR
Quercusia quercus	52	102	33.8	26.9	20	S/LR	108	306	45.4	40.8	10	S/LR
Satyrium w-album	17	1	11.0	0.3	98	DD	11	1	4.6	0.1	97	CE
Spialia sertorius	3	1	2.0	0.3	87	SU	7	1^{v}	2.9	0.1	95	EXN
Thecla betulae	25	22	16.2	5.8	64	EN	54	28	22.7	3.7	84	EN
Thymelicus acteon	–	–	–	–	–	S/LR	4	4	1.7	0.5	68	EN
Thymelicus lineola	87	359	56.5	94.7	−68	SL/R	136	628	57.1	83.7	−47	S/LR
Thymelicus sylvestris	52	165	33.8	43.5	−29	S/LR	137	288	57.6	38.4	33	S/LR
Vacciniina optilete	–	–	–	–	–	–	4	4	1.7	0.5	68	EN

4. Metapopulation dynamics in the butterfly *Hipparchia semele* changed decades before occupancy declined in the Netherlands.

*Slightly modified from: Van Strien, A.J., Van Swaay, C.A.M. & Kéry, M. (2011) Ecological Applications **21(7)** 2510–2520*

Abstract

The survival of many species in human-dominated, fragmented landscapes depends on metapopulation dynamics, i.e., on a dynamic equilibrium of extinctions and colonisations in patches of suitable habitats. To understand and predict distributional changes, knowledge of these dynamics can be essential and for this, metapopulation studies are preferably based on long-time series data from many sites, but alas, such data are very scarce. An alternative is to use opportunistic data, i.e. collected without applying standardized field methods, but these data suffer from large variations in field methods and search intensity between sites and years. Dynamic site-occupancy models offer a general approach to adjust for variable survey effort. These models extend classical metapopulation models to account for imperfect detection of species and yield estimates of the probabilities of occupancy, colonisation and survival of species at sites. By accounting for detection, they fully correct for among-year variability in search effort.
As an illustration, we fitted a dynamic site-occupancy model to 60 years of presence-absence data (more precisely, detection-nondetection) of the heathland butterfly *Hipparchia semele* in the Netherlands. Detection records were obtained from a database containing volunteer-based data from 1950-2009 and nondetection records were deduced from database records of other butterfly species. Our model revealed that metapopulation dynamics of *H. semele* had changed decades before the species' distribution began to contract. Colonisation probability had already started to decline from 1950 onwards, but this was counterbalanced by an increase in the survival of existing populations, the result of which was a stable distribution. Only from 1990 onwards survival was not sufficient to compensate for the further decrease in colonisation, and occupancy started to decline. Hence, it appears that factors acting many decades ago triggered a change in the metapopulation dynamics of this species, which ultimately led to a severe decline in occupancy that only became apparent much later. Our study emphasizes the importance of knowledge of changes in survival and colonisation of species in modern landscapes over a very long time scale. It also demonstrates the power of site-occupancy modeling to obtain important population dynamics information from databases containing opportunistic sighting records.

Introduction

Many species in human-dominated landscapes are restricted to subpopulations in patches of suitable habitat surrounded by unsuitable habitat. Finite populations may go extinct for a number of purely random causes, such as inclement weather or other stochastic effects. Only if there is inter-patch movement, unoccupied patches can be recolonised. The populations in the collection of habitat patches may thus form a metapopulation, where long-term survival is the result of a dynamic equilibrium of colonisation and extinction events (Hanski 1991). Habitat quality is one of the factors governing colonisation and extinction rates (Fleishman et al. 2002), e.g., by setting the within-patch carrying capacity or by determining the permeability of the matrix habitat surrounding the patches.

The dynamic metapopulation parameters, patch colonisation and extinction rates, are of vital importance for the long-term persistence of a species; hence, they have often been the subject of investigations. Butterflies are perhaps the classical group where a metapopulation structuring of the subpopulations in collections of habitat patches has been studied (e.g., Harrison et al. 1988, Hanski et al. 1994, Thomas et al. 1996, Wahlberg et al. 1996, Saccheri et al. 1998, Hanski et al. 2000, Hanski and Singer 2001, Baguette and Schtickzelle 2003, Davies et al. 2005, Schtickzelle et al. 2006, Bulman et al. 2007, Pellet et al. 2007, Dover and Settle 2009, Hodgson et al. 2009).

However, two serious challenges for any metapopulation study are the presence of detection error and sparse data. First, metapopulation studies hardly ever take into account the difference between real absences and nondetections. That is, in the presence of detection error, observed absences are ambiguous with respect to the occurrence status of a site and one should speak of detection/nondetection, rather than of presence/absence data (Kéry et al. 2010b). Failure to account for detection errors may lead to biased inferences on metapopulation dynamics (Moilanen 2002, MacKenzie et al. 2006) and related species distribution studies (Kéry et al. 2010b). Site-occupancy models offer the possibility to correct for this bias. They extend the classical metapopulation model to account for imperfect detection of species and yield estimates of the probabilities of occupancy, colonisation and extinction (= 1- survival; MacKenzie et al. 2006). Second, most metapopulation studies are based on data from just a few field seasons and this may corrupt estimates of metapopulation parameters (Thomas and Wilson 2002). However, long time series of standardised records of detection/nondetection data of species at many sites are very scarce.

One solution to the challenge of sparse data might be the use of opportunistic data collected in faunal and floral databases (e.g. the Dutch Butterfly Recording Database). For instance, in the Netherlands, butterflies have been studied by amateur and professional entomologists over a number of decades and many records on the occurrence of species have been collected, but often without applying standardized field protocols. Such opportunistic data suffer from large variability in field methods and search intensity among sites, which hampers deriving reliable estimates of metapopulation parameters. Recently, dynamic site-occupancy models (MacKenzie et al. 2006, Royle and Kéry 2007) have also proven useful to estimate metapopulation parameters from opportunistic data (Kéry et al. 2010a, Van Strien et al. 2010). The basic idea is that a higher observation effort implies a higher probability to detect a species, so variation in observation effort over the years can be directly translated into variation in species detectability. Records from replicate visits to a site allow estimating detection probability separately from the probability of occurrence (Kéry et al. 2010a, Van Strien et al. 2010).

If annual detection probabilities are estimated, the annual true proportion of occupied sites (occupancy) may be estimated along with annual estimates of colonisation and extinction of species at sites, corrected for all effects of changing observation effort. Hence, they appear an ideal framework for inference based on opportunistic data about the occurrence dynamics of species in fragmented landscapes.

Here we applied the dynamic site-occupancy model (MacKenzie et al. 2006, Royle and Kéry 2007) to historic data of the butterfly species *Hipparchia semele* in the Netherlands. We explored whether the model was useful to detect changes in distribution using data sampled over a time span of 60 years with greatly varying observation effort, and whether it gave insights in population dynamical processes underlying distributional changes. We tested whether the occupancy trajectory after 1990 derived from the opportunistic data was similar to that derived from an independent dataset, the Dutch butterfly monitoring scheme (Van Swaay 2005). We also produced distribution maps by plotting predicted occurrence probabilities.

Material and methods

Hipparchia semele.

Study species

We chose as study species *Hipparchia semele*, a typical heathland butterfly species.

Heathland is heavily fragmented in the Netherlands (see Dutch Environmental Data Compendium 2010). Because our study species is still widely distributed in patches of heathland on higher sandy soils in the eastern part of the Netherlands, we believe that the patches are linked by dispersal, thus together form a metapopulation in heathland areas. *H. semele* also occurs in the coastal dunes and to a much lesser extent in grasslands in other regions. Larvae feed on grasses (*Festuca* and *Agrostis* species); the adults' favourite nectar plant is *Calluna vulgaris*. *H. semele* flies late in the season compared to other butterfly species and so far no shifts in its flight period associated with climate change have been found (Van Strien et al. 2008).

Data sets analysed

To assess changes in the distribution of the species, we used two sources of data: the Dutch Butterfly Recording Database and the Dutch Butterfly Monitoring Scheme. The first scheme is a huge collection of opportunistic data, while the second scheme is a designed survey with a standardized field method.

- *Dutch Butterfly Recording Database* (filled with opportunistic data). This database comprises all historical records found in scientific journals including local 'grey' literature. In addition, records of butterfly specimens in all Dutch natural history museums and private collections were collated. Until several decades ago butterflies in the Netherlands were mainly caught for collections by a small group of entomologists. From 1980 onwards, copious new field data were collected by volunteer field workers with the aim to produce a butterfly distribution atlas (Tax 1989). In recent years sightings are made by a large group of volunteers covering many sites. The recent facilities for easy data entry on the internet have led to a new rise in the number of records, mainly through the sites www.vlindernet.nl/landkaartje, www.telmee.nl and www.waarneming.nl. All database records are validated by butterfly experts.

- *Dutch Butterfly Monitoring Scheme* (filled with monitoring data)
  This scheme runs from 1990 onwards and applies the method developed for the British Butterfly Monitoring Scheme (Pollard and Yates 1993). Counts are conducted along fixed transects of about 1 km, consisting of smaller sections, each within a homogeneous habitat type. Transects were mainly chosen by free choice of observers. Volunteers record all butterflies 2.5 m on both sides and 5 m ahead of and above them. Weekly surveys are conducted between 1 April and 30 September when weather conditions meet specified criteria (Van Swaay 2005). Most transects are recorded by skilled volunteers, and their sightings are validated by experts.

Definition of occupancy
Occupancy is a species' probability to occur at a site during the species-specific flight period. Many old records in the databank were stored at a 5 x 5 km resolution and hence, we use 5 x 5 km squares as our definition of a site. We estimated annual occupancy, i.e., the proportion of occupied sites in the statistical population represented by our sampled squares. We restricted the analysis to the potential range of *H. semele*, defined here as those 566 sites in which observations had ever been made since 1950. Given the sampling intensity over the years and the dependence of the species on specific habitat types, we believe it unlikely that the species has ever occurred at a site from which it had never been reported. All records from the database (typically counts) were aggregated to detection records per site. Counts derived from the monitoring scheme were also quantized into detection/nondetection data at a 5 x 5 km resolution.

Deducing nondetection records
Nondetection data for *H. semele* were easily extracted from the Dutch Butterfly Monitoring Scheme; they were simply all visits made without any recorded sightings of *H. semele* within the sites of its range. It was less straightforward for the Recording Database, because that was not based on a standardized field protocol. We deduced nondetection records from the sightings of all other butterfly species in the database (Kéry et al. 2010a): any observation of *H. semele* on a particular day and site (5 x 5 km square) was taken as a detection (1) in the dataset, and an observation of any other species within the flight period of *H. semele* was taken as a nondetection (0). Fieldworkers may not record all species observed, hence like in Van Strien et al. (2010), we distinguished three categories of data quality in these data, and accordingly, extracted three data sets from the database containing opportunistic data: (1) single-records data, (2) short daily species lists and (3) comprehensive daily species lists. Single records data formed the lowest-quality data and were defined as records of one species made by a single observer on a single day at a single site. Such data are usually coincidental observations and are predominant in museum collection data (McCarthy 1998). We called all reports of 2 or 3 species at one site and day by one observer short daily species lists because in the Netherlands, > 3 species are generally seen during field trips within the flight period of *H. semele* (Van Swaay, pers. obs.). Reports of more than 3 species formed the dataset of comprehensive lists. Obviously, in the first two data-quality categories, many zeroes are in fact nonreported observations of the study species. But most comprehensive species lists (87%) contained records of one or several of the most common ('uninteresting') butterfly species such as *Pieris rapae, Pieris napi, Maniola jurtina* or *Coenonympha pamphilus*. Because *H. semele* is generally regarded as a more 'interesting' species by observers, we believe that in comprehensive species lists *H. semele* would have been recorded when detected.

The set of opportunistic detection/nondetection data extracted from the Dutch Butterfly Recording Database for *H. semele* contained about 9 000 detections and 106,000 nondetections for all three data quality categories and all 566 sites together. The nondetection records are a mixture of real absences, nondetections and detected presences that failed to be reported. During 1950-1980 about 100 sites were annually visited during the flight period and within the range of this species with on average a 2-3-fold replication of visits (Figure 4.1). Many of these data were single-records while comprehensive daily species lists were few in this period. The number of records in the database with opportunistic data increased after 1980, and especially after 1990. Nowadays, around 6,000 records are collected each year with up to 30 replicate visits per site and year, including many comprehensive daily species lists. The number of sites in the Dutch Butterfly Monitoring Scheme was considerably lower: annually about 60 5 x 5 km sites were surveyed within the range of this species, with on average 6.6 replicated visits within the flight period of *H. semele*.
To use these data for inferences about the species in the Netherlands, trends must be representative for the entire range of the species across the whole study period. In the opportunistic data, the main regions of the country in which the species occurs appeared to have been surveyed every year more or less in proportion to their occupied areas, with little variation among years. On average (1950–2009), 61.8 ± 0.4% of all squares surveyed per year were on sandy soils, 3.9 ± 0.2% in coastal dunes and 34.2 ± 0.3% in other areas. These values are close to the distribution of all 566 sites from which the species had ever been reported (60.8%, 6.5% and 32.6% for these three regions respectively), hence we assume that the annual surveys do not deviate substantially from random sampling. In the monitoring scheme, the free choice of observers to select transects have led to oversampling of coastal dunes as compared to the other regions, thus trends in occupancy were not representative for the Netherlands as a whole. For higher sandy soils only however, the monitoring data do not deviate much from random sampling. Comparison of trends in occupancy between monitoring data and opportunistic data were therefore limited to higher sandy soils.

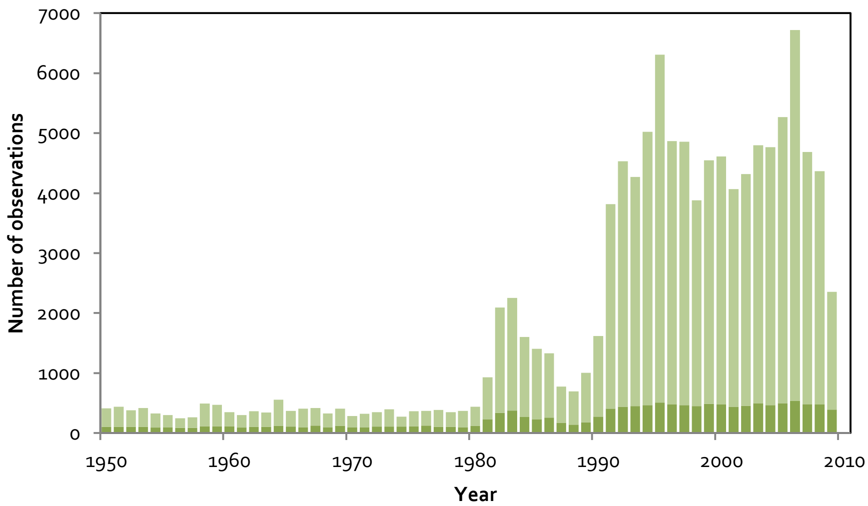

Figure 4.1: Number of opportunistic detection/nondetection observations per year at all 566 5 x 5 km² sites within the Dutch range of* Hipparchia semele. *Dark bars represent first observations and light bars represent replicate visits observations. Dark bars thereby also represent the number of sites surveyed annually, ranging from 15% in 1956 to 95% in 2006.
The amount of data in 2009 is lower because not all collected data were already available.

Statistical analysis

We used the dynamic site-occupancy model (MacKenzie et al. 2006) as described in WinBUGS code by Royle and Kéry (2007) and Royle and Dorazio (2008, p. 309) and as applied to opportunistic data by Kéry et al. (2010a) and Van Strien et al. (2010), to estimate annual occupancy probability (ψ) and its dynamic components (survival probability (φ) and colonisation probability (γ)), adjusted for detection probability p. Estimating p is only possible if repeated visits are available for at least some sites within a season (MacKenzie et al. 2006). Site-occupancy models therefore require replicated detection/nondetection data collected on a number of sites that are arranged in so-called detection histories per site during a single season. An example is "010" for a study species detected during the second visit, but not during the first and third visit to a site in a single period. The replicated surveys need to be done within a period of closure. Closure means that a site must stay either occupied or not but must not become permanently abandoned or colonised during the period of surveys within a 'season' (usually, a year). To meet the closure assumption, we restricted the data to the known flight period of *H. semele*, Julian dates 176 – 263 (25 June and 20 September), and discarded some more extreme dates with sightings (Bos et al. 2006).

Both components of the model (i.e., occupancy/colonisation/extinction and detection probability p) may be formulated as a function of covariates, but here we only used covariates for detection. Detection of butterflies varies over the season mainly due to a changing number of adult butterflies over the course of a flight period (Pellet 2008). Hence, we used the Julian date as a covariate for p. In addition, data quality was used as a categorical covariate for p in the opportunistic data set. Effects of both covariates were included in the model via a logit link:

$$\begin{aligned} \text{logit}(p_{ijk}) = \alpha_k + \beta_1 * \text{date}_{ij} + \beta_2 * \text{date}_{ij}^2 \\ + \delta_1 * (\text{data quality category 2})_{ij} \\ + \delta_2 * (\text{data quality category 3})_{ij} \end{aligned}$$

where p_{ijk} is the probability to detect the species at site i during visit j in year k, α_k is the annual intercept, β_1 and β_2 are the linear and quadratic effects of the date of visit j at site $_i$ and δ_1 and δ_2 are the effects of data quality category 2 and 3, relative to data quality 1. The intercept α_k was estimated as a random year effect (see Kéry 2010 for examples of WinBUGS code for random effects).

We fitted the models in a Bayesian mode of inference using JAGS (Plummer 2009) on the computer cluster LISA (https://subtrac.sara.nl), with essentially the same WinBUGS code as described by Royle and Dorazio (2008). We chose conventional vague priors for all parameters, i.e., uniform distributions between 0 and 1 for all parameters except α_k (U(-5, 5)) and β_1, β_2, δ_1 and δ_2 (U(-10, 10)). Parameter estimates (posterior means) were robust to changes in prior specifications, except for the effects of date, β_1 and β_2, which had some influence on the estimated relation between detection probability and Julian date. However, there was hardly any influence on the estimates of all other parameters, such as occupancy.

For each analysis, we ran three Markov chains with 6,000 iterations each and discarded the first half as burn-in. These specifications were sufficient to achieve convergence based on the Gelman-Rubin Rhat statistic (Rhat <1.1). We were interested in the actual set of studied sites; hence, we used the finite-population occupancy estimate, which is estimated more precisely than the occupancy in an infinite population of sites (Royle and Kéry 2007). Linear trends in occupancy ψ, colonisation γ, survival φ and detection probability p across years were estimated as derived parameters within the model, both for the entire 60-year period as for

1950-1969, 1970-1989 and 1990-2009. These 20-year periods were chosen to enable comparison with monitoring data (1990-2009).
We report posterior means and standard deviations as point and uncertainty estimators of a parameter. For comparison, we also computed the naïve occupancy estimates, i.e., without taking into account *p* using a simple logistic regression analysis on observed occurrence per site per year. Our model predicted the probability of occurrence of the study species for each site and year, including years without visits to a site. When plotting these values in annual distribution maps, we treated probabilities ≥ 0.5 as presences and < 0.5 as absences.

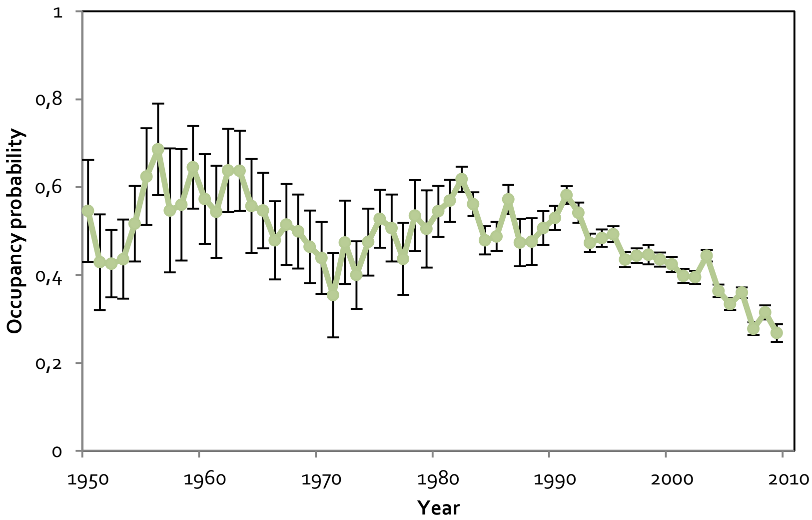

Figure 2. Annual site occupancy probability (± se) of* Hipparchia semele *based on the dynamic site-occupancy model using opportunistic data. Sites are 5 x 5 km squares.

Results

Naïve occupancy estimates based on opportunistic data, without taking into account *p*, increased between 1950 and 2009 (logistic regression trend: 0.003 ± 0.0015; P<0.05). In sharp contrast, the site-occupancy model showed no change in occupancy until 1990, whereafter a considerable decline occurred (Figure 4.2; Table 4.1). The decline since 1990 was confirmed by the independent monitoring data (for sandy soils only: trend in ψ -0.016 ± 0.003 and -0.017 ± 0.001 for monitoring and opportunistic data respectively). Our model suggests that the downward trend in occupancy since 1990 was due to a decline in the colonization probability of unoccupied sites (Table 4.1), a finding which was again confirmed by the monitoring data (for sandy soils only: trend in γ -0.013 ± 0.004 and -0.006 ± 0.002 for monitoring and opportunistic data respectively). Survival rates after 1990 have not changed significantly (Table 4.1) (for sandy soils only: trend in φ 0.004 ± 0.005 and -0.003 ± 0.002 for monitoring and opportunistic data respectively). Remarkably, however, the trajectories of colonisation and survival probabilities indicated that metapopulation dynamics of this species in the Netherlands has started to change long before 1990 (Figure 4.3; Table 4.1). Colonisation rates appeared to have declined steadily from the 1960s onwards. Interestingly, there was an increase in survival in 1970-1989, but survival did not grow thereafter and stabilized at a level slightly lower than in 1970-1989 (Table 4.1). Thus, until 1990 the decline in colonisation was counterbalanced by a rising annual survival of populations, resulting in a dynamic equilibrium of occupancy (MacKenzie et al. 2006). The rise in survival until 1990 is not merely the consequence of the contraction of the species range to better sites, because then occupancy would have declined too. Instead, at some sites survival must have improved in 1970-1989.

*Table 4.1: Parameter estimates (posterior means and standard deviations) under a dynamic site-occupancy model fitted to Hipparchia semele opportunistic data from the Netherlands. No estimates of Julian date and data quality are available for the separate 20-year periods, because model run was for the entire 60-year period. Trend is detection p_j refers to trend in annual detection probability per 5 x 5 km² site in mid August (day 225) for the highest data quality category (see also Figure 6). * P<0.05 as derived from Bayesian credibility intervals.*

	1950-1969	1970-1989	1990-2009	1950-2009
trend in occupancy ψ_j	0.001 ± 0.005	0.005 ± 0.003	-0.014 ± 0.001 *	0.001 ± 0.005
trend in colonisation γ_j	-0.006 ± 0.006	-0.006 ± 0.003 *	-0.006 ± 0.001 *	-0.006 ± 0.006
trend in survival φ_j	0.004 ± 0.007	0.009 ± 0.005 *	-0.001 ± 0.002	0.004 ± 0.007
trend in detection p_j	0.000 ± 0.001	-0.004 ± 0.001 *	0.003 ± 0.003	-0.001 ± 0.001
Julian date effect β_1	not available	not available	not available	9.521 ± 0.345
Julian date effect β_2	not available	not available	not available	-9.626 ± 0.343
data quality effect δ_1	not available	not available	not available	1.120 ± 0.126
data quality effect δ_2	not available	not available	not available	2.536 ± 0.185

This idea was confirmed in a further analysis in which we separately estimated colonisation and survival rates for subsets of sites differing in the area of suitable habitat. For all sites in higher sandy soils we derived the area of heathland from Anonymus (1987), reflecting the situation in ca. 1980. In the subset with the largest area of suitable habitat i.e. with > 200 ha of heathland per site, survival has indeed increased prior to 1990 (Table 4.2). The same has happened in the subset of sites with 20-200 ha of heathland, whereas in less suitable sites with < 20 ha of heathland survival has not improved before 1990. After 1990 survival did not increase in any of the three subsets and even declined significantly in the subset with sites containing 20-200 ha heathland (Table 4.2). Colonisation has declined in 1950-2009 in sites with <20 and 20-200 ha heathland, but not in sites with > 200 ha heathland. There were no differences in occupancy between the subsets in 1960-1970 (presumably because at that time the area of heathland differed not as much between the subsets as in ca. 1980), but occupancy has declined strongly in sites with < 20 ha heathland and increased in sites with > 200 ha heathland (Figure 4.4).

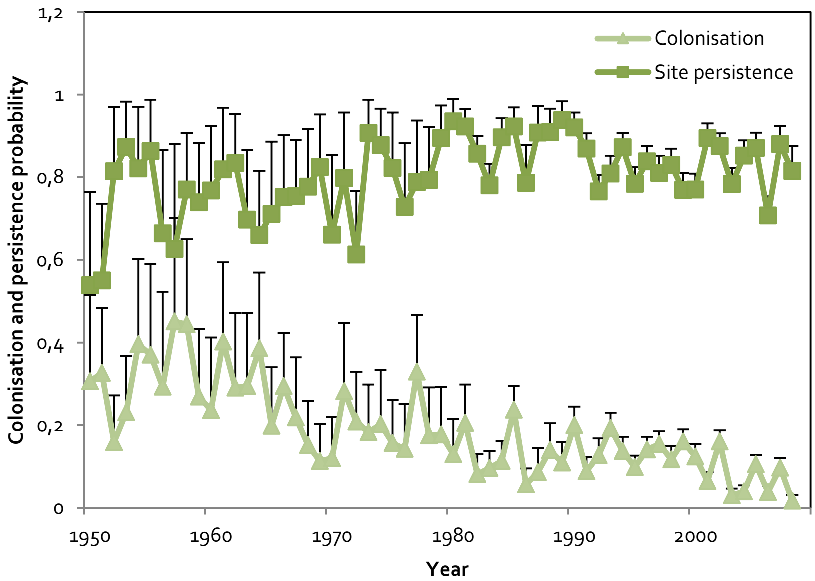

***Figure 4.3:** Annual rates of site colonization (triangles) and site persistence (squares) (+ se) of* Hipparchia semele *analyzed with a dynamic site-occupancy model using opportunistic data. Sites are 5x 5 km squares.*

Table 4.2: Trends in occupancy, colonization, and survival (±SE) for three subsets of opportunistic data with different surface areas of heathland.
***Notes:** All sites were 5x5 km squares situated in the region with higher sandy soils. The results were based on separate analyses for each data set.*
** P < 0.05 as derived from Bayesian credibility intervals.*

Heathland sites and trends	1950-1969	1970-1989	1990-2009	1950-2009
Heathland <20 ha (n=55 sites)				
trend in occupancy ψ_j	-0.001 ± 0.008	-0.003 ± 0.008	-0.021 ± 0.005 *	-0.007 ± 0.001 *
trend in colonisation γ_j	-0.004 ± 0.010	-0.004 ± 0.010	-0.013 ± 0.006 *	-0.008 ± 0.002 *
trend in survival φ_j	-0.001 ± 0.011	-0.004 ± 0.010	-0.006 ± 0.009	-0.001 ± 0.002
Heathland 20–200 ha (n=90 sites)				
trend in occupancy ψ_j	-0.004 ± 0.007	0.003 ± 0.006	-0.019 ± 0.003 *	-0.003 ± 0.001 *
trend in colonisation γ_j	0.000 ± 0.010	-0.018 ± 0.009 *	-0.002 ± 0.004	-0.010 ± 0.001 *
trend in survival φ_j	0.000 ± 0.011	0.023 ± 0.008 *	-0.011 ± 0.005 *	0.006 ± 0.002 *
Heathland >200 ha (n=49 sites)				
trend in occupancy ψ_j	0.014 ± 0.007 *	0.019 ± 0.007 *	-0.008 ± 0.002 *	0.005 ± 0.001 *
trend in colonisation γ_j	0.005 ± 0.010	0.013 ± 0.009	-0.008 ± 0.009	-0.002 ± 0.002
trend in survival φ_j	0.011 ± 0.010	0.023 ± 0.009 *	-0.003 ± 0.003	0.007 ± 0.002 *

We expected changes in metapopulation dynamics to happen more frequently in areas outside the core areas that the species inhabits (Hanski 1991, Dover and Settele 2009). Indeed, the distribution of *H. semele* in 2009 was much more concentrated in several core areas than in 1990; hence, the decline seemed to have occurred to a considerable extent in areas outside core areas (Figure 4.5a-b). Summarizing, over the entire period, the distribution of *H. semele* in the Netherlands is getting more and more concentrated in a few core areas with large areas with suitable habitat, while the species has disappeared from many areas with marginally suitable habitat. Until 1990, however, this process remained obscure, because the increased survival of the species in more suitable sites compensated for the loss of marginal sites.
All metapopulation parameters estimated in this study were adjusted for detection probability. The detection probability *p* of *H. semele* was lower at the start and the end of the flight period and had a peak around day 225. Detection probability was especially low in single-records data (Figure 4.6), which indicates considerable underreporting of the species after detection during these visits. Detection probability was higher in short daily species lists and considerable higher in comprehensive species lists. The probability to detect *H. semele* dropped slightly after 1970 and increased again after 1990 (Table 4.1; figure 4.6).

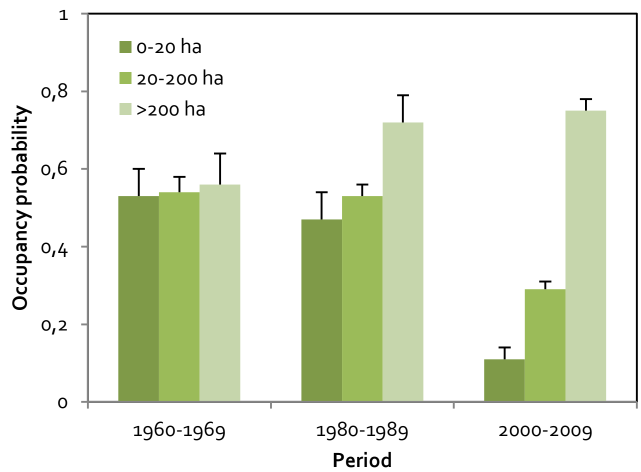

***Figure 4.4:** Mean annual occupancy (± se) per decade of* Hipparchia semele *in 5 x 5 km sites on sandy soils with different surface areas of heathland based on opportunistic data.*

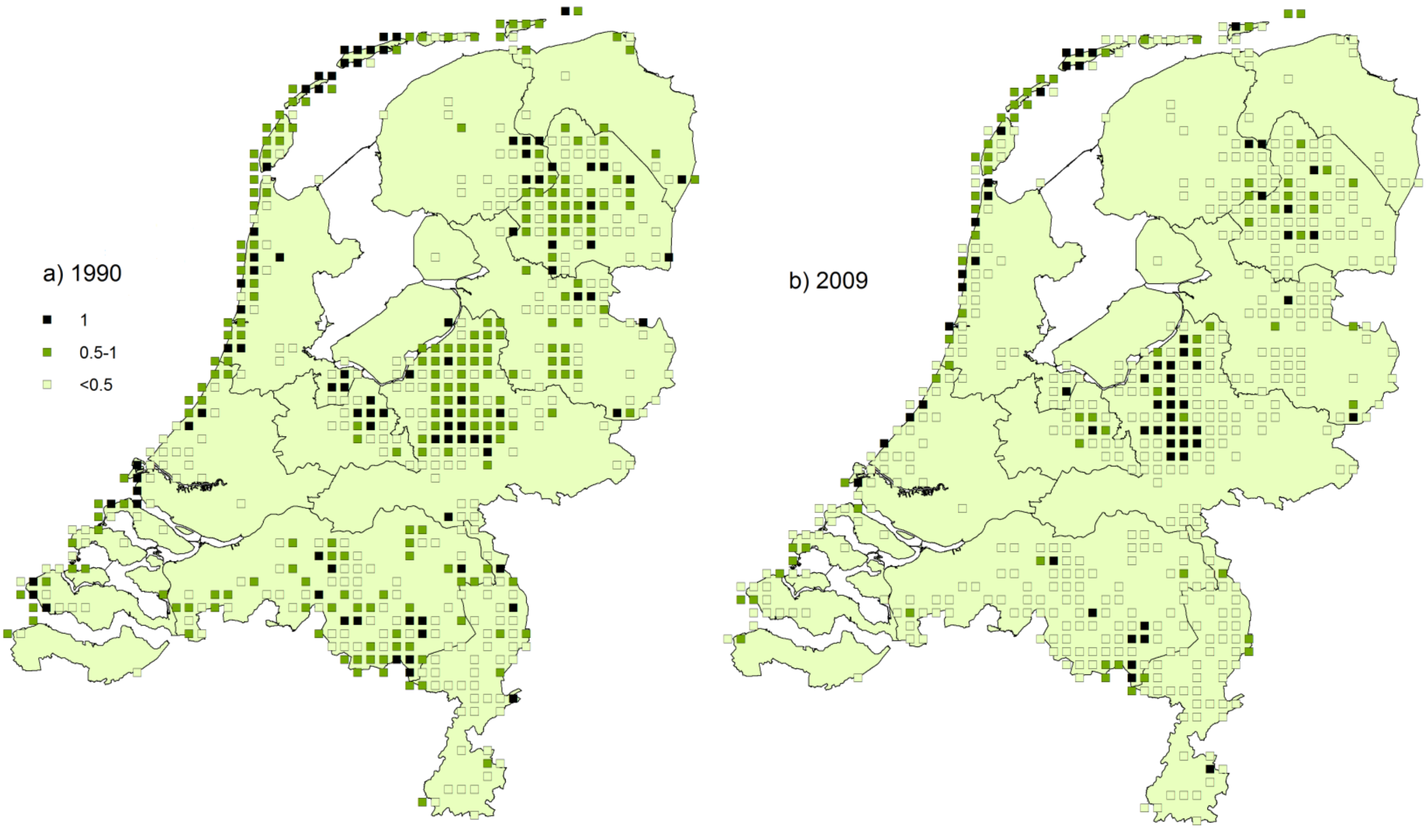

Figure 4.5: Distribution of* Hipparchia semele *in (A) 1990 and(B) 2009 estimated under a dynamic site-occupancy model, all based on opportunistic data. The symbols denote different levels of occupancy probability per site. Recorded sightings result in an estimate of occupancy probability of 1. Areas without a symbol were never occupied in 1950–2009. Note that the percentage of occupied sites in 1990 and 2009 is similar to the occupancy in these years as presented in Figure 2.

Discussion

Metapopulation dynamics

We found that the metapopulation dynamics in the butterfly *H. semele* in the Netherlands had changed several decades before a decline became visible in the proportion of occupied sites. Though a reduced colonisation rate in later years was not unexpected, the steady and strong decline of the colonisation rate during a period of 50-60 years has not been reported in earlier studies of this species (Van Swaay 1990, Bos et al. 2006). The decline in colonisation is probably due the loss of suitable habitat in heathland areas as well as a decline in connectivity of these patches. Because sheep grazing intensity was lowered and nitrogen load from the air has increased after 1950, the succession rate has been speeding up, leading to many heathlands overgrown with grasses and shrubs which no longer had a structure appropriate for *H. semele* (Bos et al. 2006). The ongoing succession also led to the conversion of many heathlands into woodland (Diemont 1996, Dutch Environmental Data Compendium 2010). In sites with large areas of heathland, suitable habitat will not easily disappear completely as in areas with small and scattered patches of heathland, also because management is usually better maintained in larger heath patches (Van Swaay, pers. obs.). The bigger decline in occupancy in recent years in sites further away from core areas suggests that increased isolation of heathland contributed to reduced colonisation rate, as expected in metapopulation theory (Hanski 1991). Dennis et al. (1998) also reported that both area and isolation were associated with patterns of presence and absence of *H. semele* on British and Irish offshore islands. The increased survival of the species in 1970-1990 has completely escaped previous attention and we can only speculate why survival has improved. Survival is best predicted as a function of local population size (Pellet et al. 2007), so it is likely that population size has increased in that period. Perhaps the species has benefitted from the cool and rainy weather in this period, or the rising nitrogen deposition temporarily increased the survival of the larvae on these extremely poor soils. It requires more extended models to elucidate this further. In future modelling efforts, area of suitable habitat, distance to core area, management and other variables may be applied as covariates for first-year occupancy or for the colonisation and survival parameters (Royle and Dorazio 2008). This enables to examine the effects of covariates on colonisation and survival rates directly.

Our results also provide a direct, mechanistic evidence for an extinction debt (Kuussaari et al. 2009). Arguably, land use changes have been depressing patch colonisation probabilities as early as in the 1960s, and yet, the results of this only became evident many decades afterwards. There are very few examples of direct measurements of extinction debts over such long time spans (e.g. Polus et al. 2007) and most assessments of extinction debts have so far been implied indirectly (Kuussaari et al. 2009), e.g., by regressing current patterns of occurrence of single species, or of species counts, on past values of land use and patch characteristics (Findlay and Bourdages 2000, Hawbaker et al. 2006, Piha et al. 2007, Sang et al. 2010). The general lack of appropriate high-quality historical data is considered a key limiting factor for studying extinction debt (Kuussaari et al. 2009), but here we have shown that opportunistic data can be a useful surrogate.

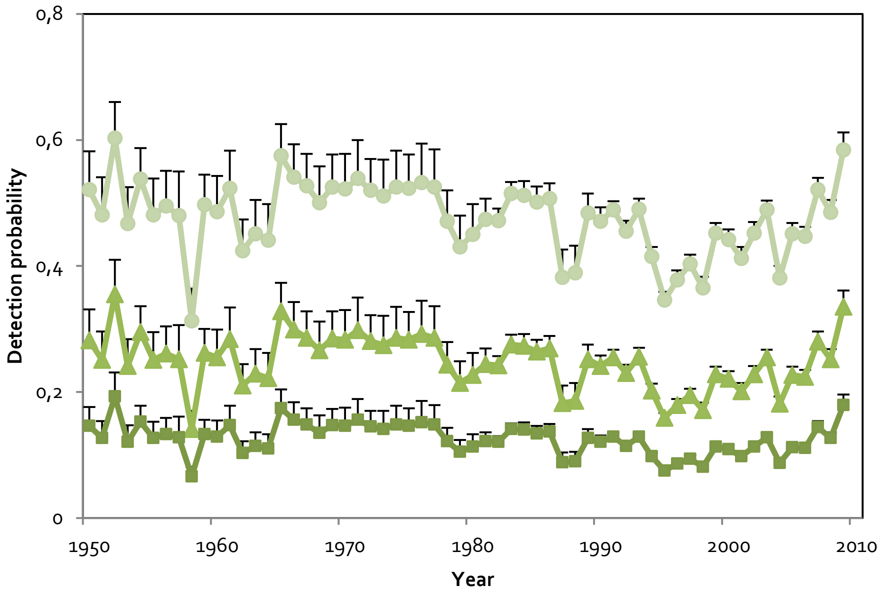

Figure 4.6: Annual detection probability (+ se) per 5x 5 km site of* Hipparchia semele *in mid-August (day 225) per data qualitycategory in opportunistic data. The lower line refers to the lowest data quality and the top line to the highest quality category.

Estimation of trend and distribution

The finding of a peak in detection probability around day 225 corresponds to the literature (e.g. Bos et al. 2006) and reflects the peak in the seasonal abundance in sites. The drop in probability around 1980 to detect *H. semele* is perhaps caused by the recruitment of a large number of relatively inexperienced field observers for the atlas project that started in that period (Tax, 1989). Although the trend in *p* is not significant in 1990-2009, we see an apparent increase in recent years (Figure 4.7), probably from the fact that observers collecting opportunistic data increasingly use information available on the internet to direct them to sites where interesting species such as *H. semele* have been spotted recently. Naïve occupancy

estimates, without taking into account p, may therefore be expected to decline even more than based on the site-occupancy model. Yet, we found the contrary: the naïve trend increased, rather than declined, between 1950 and 2009. This is because the consequences of $p < 1$ for occupancy estimation do not only depend on the value of p for a single visit, but also on the number of visits (Kéry et al. 2010a). The number of replicated visits has increased much after 1980 (Figure 4.1) and this not only counterbalanced lower detection probabilities, but even led to an artefactual increase in occupancy because of the increased probability to detect a species at least once during the season. An artefactual increase in naïve occupancy estimates is typically found in most collections of opportunistic data, where not only the number of sites surveyed has increased over time, but sites were also investigated more frequently and often more thoroughly. This makes it difficult to separate changes in distribution from changes in observation effort (Dennis et al. 1999, Dennis and Thomas 2000). Many attempts have been made to adjust the opportunistic data for differences in observation effort (for an overview see Telfer et al. 2002), but these were merely *ad hoc* approaches. Dynamic site-occupancy models offer a more general and comprehensive approach to adjust for unequal observation effort that is firmly based on sampling theory. These models achieve a "mechanistic" correction for the effects of varying observation effort (MacKenzie et al. 2006, Royle and Kéry 2007, Royle and Dorazio 2008), unlike some other approaches that fit proxies for observation effort as a covariate, e.g. based on the number of records of abundant species that are assumed not to have declined.
In addition, site-occupancy models may produce annual distribution maps from opportunistic data. Though the making of distribution maps has much improved in recent years (see e.g. Elith et al. 2006), extremely few maps are adjusted for differences in detection probabilities between sites (Royle et al. 2005, Kéry et al. 2010b). Estimates of distributional changes are easily corrupted if all sites without sighting are treated as real absences, because almost certainly a part of these absences are in fact non-detections of real presences (Kéry et al. 2010b). Site-occupancy models enable to take this into account in a subtle way. For our annual maps of *H. semele*, our model predicted presences and absences per site per year from the detection probability estimated per visit and year, the number of visits to the site, taking into account Julian date and data quality of the visit, the presence or absence in the preceding year as well as the annual colonisation and survival rates.

Perspectives

Dynamic site-occupancy models act like a "currency-converter" for the data when comparing opportunistic data over time and enable to produce reliable occupancy trend estimates from databases containing opportunistic observation data (Altwegg et al. 2008, Kéry et al. 2010a, b, Van Strien et al. 2010). Consequently, as demonstrated here, site-occupancy models offer new perspectives to derive inferences on trends and distribution from old detection/nondetection data (Tingley and Beissinger 2009). That will be particularly beneficial for species that like butterflies have been much recorded in former days and which have been collected as species lists rather than as single records data (Van Strien et al. 2010). The perspectives of site-occupancy models are even higher for future data, because a rapidly increasing amount of detection/nondetection data is currently being collected in the framework of citizen science projects.
Nevertheless, occupancy models may suffer from biases if their assumptions are violated. We discuss several of the key assumptions here and refer to MacKenzie et al. (2006) for a more extensive discussion of occupancy model assumptions. Firstly, although we restricted the data to meet the closure assumption (see methods), we cannot be sure that all sites were permanently occupied during the

period selected. But some lack of closure is not fatal, for instance, when animals randomly move in or out of occupied sites. Such random lack-of-closure will simply reduce the detection probability and does not bias the occupancy parameter, although the latter must be interpreted as probability of use rather than probability of permanent occurrence (MacKenzie et al. 2006). A related problem may reside in the fact that different records in the same spatial sample unit (i.e., 5 x 5 km in our study) may refer to vastly different sites, so that the non-detection of species A may not be informative about detection probability of that species because a different place was surveyed from the one that may have produced a positive detection of A at some other time. Kendall & White (2009) demonstrated that especially sampling of spatial subunits within a site leads to bias in occupancy estimates, but not sampling with replacement. We believe that the collection of opportunistic data by many observers is comparable to sampling with replacement rather than to sampling without replacement.
Furthermore, our method depends on the definition of a nondetection event. We deduced nondetection records from the sightings of other butterfly species (see methods). Our procedure generated many nondetection records for every single detection record, but these records may not be as informative about nondetections of other species as we supposed. To a lesser degree this also holds for the short and comprehensive daily species lists. Van Strien et al. (2010) tested the same procedure using dragonflies and found similar occupancy trends in opportunistic data as in independent monitoring data, thereby suggesting that the procedure worked appropriately. But the robustness of the procedure to deduce nondetection data requires further investigation, possibly by simulation studies, In addition, surveys without the detection of any species at all, or only of common and therefore less 'interesting' ones, are most probably under reported in the data. This might lead to a proportion of zeroes that go missing. If this proportion of "missing zeroes" changes over time, one might fear that biases are introduced into the estimates of occupancy, colonisation and extinction. However, simulations reported in Kery et al. (2010a) suggest that the method is surprisingly robust to the latter kind of error. Finally, site-occupancy-models assumed the absence of any site-dependent heterogeneity in detection (MacKenzie et al. 2006). Unmodelled detection heterogeneity leads to underestimation of occupancy in these types of models (Dorazio 2007). Moreover, if there is detection heterogeneity and its magnitude varies by year, this bias could vary by year also. A main source of heterogeneity in detection is site-specific abundance. Therefore, this type of bias can be reduced by taking into account site-specific covariates related to site-specific abundance, in our case e.g. the area of heathland per site.
In conclusion, we believe that the fact that the parameters of metapopulation dynamics can now be directly studied over long time periods using opportunistic faunal or floral data and using dynamic site-occupancy models opens up new opportunities in ecological research and applications. But addressing assumptions of occupancy models remains essential to drawing valid inferences.

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Part II: Monitoring trends in butterfly abundance

5. Butterfly monitoring in Europe: Methods, applications and perspectives

Slightly modified from:
Van Swaay, C.A.M., Nowicki, P., Settele, J., Van Strien, A.J. (2008)
*Biodiversity and Conservation, **17 (14)**, 3455-3469.*

Abstract

Since the first Butterfly Monitoring Scheme in the UK started in the mid-1970s, butterfly monitoring in Europe has developed in more than 10 European countries. These schemes are aimed to assess regional and national trends in butterfly abundance per species. We discuss strengths and weaknesses of methods used in these schemes and give examples of applications of the data. A new development is to establish supra-national trends per species and multispecies indicators. Such indicators enable to report against the target to halt biodiversity loss by 2010. Our preliminary European Grassland Butterfly Indicator shows a decline of 50% of the population indexes of the characteristic indicator species between 1990-2005. We expect to develop a Grassland Butterfly Indicator with an improved coverage across European countries. We see also good perspectives to develop a supra-national indicator for climate change as well as an indicator for woodland butterflies.

Lycaena dispar *is a species mentioned on the annexes of the Habitats Directive. The population trend of this species has to be reported in every Member State to the European Union.*

Introduction

Insects are by far the most species-rich group of animals, representing over 50% of the world's biodiversity (May 1988; Gaston 1991; Groombridge 1992). Contrary to most other groups of insects, butterflies are well-documented, easy to recognize and popular with the general public (De Heer et al. 2005; Thomas 2005). Many European butterflies have decreased considerably in abundance in recent years (Van Swaay et al. 2006). As a result, nowadays 71 out of the 576 European butterfly species are considered as threatened in Europe (Van Swaay and Warren 1999). The decline in abundance of butterfly species has largely been assessed by using distributional data to examine the change in their area of distribution (Van Swaay 1990, Maes and Van Swaay 1997; Telfer et al 2002). But this approach has several shortcomings. First, it underestimates the rate of population decline because generally species decrease in population numbers first before they disappear locally and regionally (Thomas and Abery 1995). Secondly, most available distributional data suffer from differences in sampling effort over time, which makes it difficult to separate changes in distribution from changes in sampling effort (Dennis et al. 1999). Reliable estimates of trends can only be based on long series of distributional data, because only then correction for sampling effort is possible (Van Swaay et al. 1990, Maes and Van Swaay 1997; Telfer 2002), but even then the results should be treated with caution. In order to get early warning signals, it is better to assess trends in population numbers based on monitoring schemes with standardized sampling efforts. These were the reasons for setting up a national butterfly monitoring scheme in the UK in 1976 (Pollard 1977). This has inspired many others and the number of schemes has gradually increased in Europe (Table 5.1; Figure 5.1; see Kühn et al. 2005, and contributions therein). New schemes are being planned, e.g. in Denmark and Sweden. The number of transects differs much between the current schemes, ranging from just a few transects per country to several hundreds in the UK and the Netherlands. In 2004 Butterfly Conservation Europe (www.bc-europe.eu) was founded and had an important role in bringing together and co-ordinating work on butterfly monitoring in Europe.

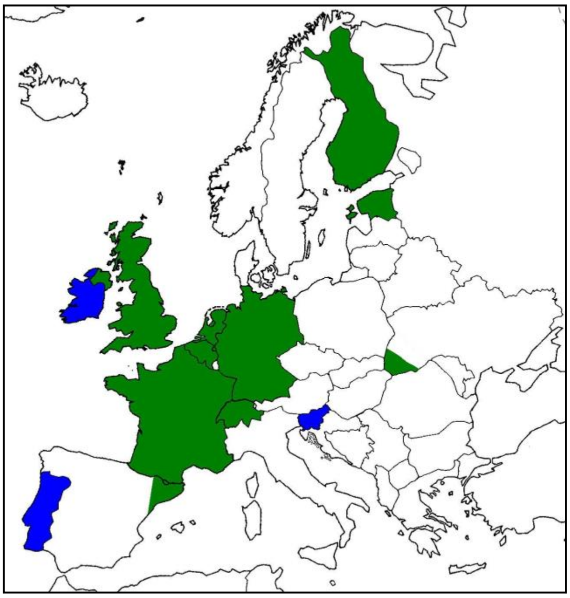

Figure 5.1: Location of Butterfly Monitoring Schemes in Europe in 2007 (green shading - active schemes, blue – planned schemes).

In this chapter we describe the main methods used in the current schemes and give a few examples of applications of the data. We discuss the use of butterflies in biodiversity indicators and the perspectives of European butterfly monitoring and indicators.

Table 5.1: Active Butterfly Monitoring Schemes in Europe in 2007.
*The data from countries or regions marked by * were used for the Grassland Indicator (the first European Butterfly Indicator).*
[1] *only for* Maculinea nausithous, M. teleius *and* Lycaena dispar *(Settele 1998)*
[2] *including Northrhine-Westfalia (Kühn et al. 2008; but excluding the Pfalz region, from where* Maculinea nausithous *monitoring data of Settele (1998) were used specifically for the grassland indicator)*

Butterfly Monitoring Scheme	Year established	No. sites in recent years
United Kingdom*	1976	600
Transcarpathia (Ukraine)*	1983	20-30
Germany (Pfalz region)*[1]	1989	100
The Netherlands*	1990	700
Belgium (Flanders)*	1991	10-20
Spain (Catalunya)*	1994	50-60
Switzerland (Aargau)*	1998	100+
Finland*	1999	100
Switzerland	2000	100+
Germany (Northrhine-Westfalia)*	2001	100
France (Doubs and Dordogne)*	2001	10
Jersey (Channel Islands)	2004	25
Estonia	2004	7
Germany (entire country)	2005	450[2]
France (entire country)	2005	75
Slovenia	2006	30
Ireland	2007	Not clear yet


In many countries in Europe, Maniola jurtina *is the most abundant butterfly on the monitoring transects.*

Table 5.2: The main characteristics of the 'Traditional' and 'Reduced effort' Butterfly Monitoring Schemes (based on Roy et al. 2005, 2007; Heliölä & Kuussaari 2005; Van Swaay 2007)

	Traditional BMS	Reduced effort BMS
Characteristics	Based on weekly counts, mostly with free choice of site	Based on a higher number of transects, counted only a few times a year, on random or pre-selected sites
Objectives	• National, regional and local indices and trends • Possibility to compare local indices and trends with regional or local trends • Can be used to evaluate nature conservation measures • Research e.g. climate change	• National indices and trends for widespread species or targeted at individual rare species
Common features	• Transects should be as far as possible representative for the sampling unit (e.g. of a site, species flight area) • Transects should preferably be in one 'rough' habitat type (like grassland, woodland, heathland, etc.), to enable trends by habitat to be more easily assessed - relevant to potential future EU analyses. • Length of transect: no prescribed limit but for practical reasons it is best if a transect walk takes 15-60 minutes, and travel time to the site is not more than 15-30 minutes. That will reduce the length of a transect mostly to a maximum of two kilometres. • Length of sections: can vary or be fixed. In case of a fixed length, 50m has proven to be a practical length. • Transect width: preferably 2.5 m on each side (5 m width). • Sections should preferably be homogeneous according to habitat type, because this allows for weighting by habitat type when calculating indices and trends. Weighting improves the quality of the results. However, because of succession, urbanisation, etc, sections may become heterogeneous in time. This may lead to a situation where a section contains several habitat types. Therefore the habitat type of a section should be established regularly (at 5 or 10 yearly intervals). • Habitat classification: preferably cross referenced to EUNIS. • Time frame during the day. General between 10 h and 17 h, preferably always during the same part of the day, sticking to this over the years. • Transects should only be walked when butterflies are fully active (i.e. under suitable weather conditions: temperature above 17°C, or 13–17 °C in sunny weather, wind less than 6 Beaufort and no rain). • Lumping of species (e.g. Blues). In some cases there is no alternative. But take care that if the recorder starts to discriminate between the species, you should put all earlier years to 'missing value'. • Should each transect be recorded each year? This is not necessary, although trend calculations will improve if some transects are counted annually. • In case of a lack in volunteers/resources, it is more effective and gives better trends, if many transects are counted (though not each year), than a few transects which are counted annually (e.g.: if 30 transects can be counted each year, it is better to count these every three years, so in total 90 transects are counted on a three year basis, than the 30 identical transects counted each year). However, trend calculations improve even more if a few of these transects are counted annually.	
Differences	• Number of counts: preferably each week covering the flight periods of all species being monitored. Weekly counts offer the opportunity for extra assessments, but if the objective is only to produce national trends then the effort can be reduced, but never to less than twice a month.	• Number of counts: 3-5 annually (e.g. one each month, like in France, or three visits in July/August, like in the proposed wider-countryside BMS in UK) but with more transects. Visits should be targeted to the period in which you expect to collect most information. Maintain a level of flexibility.

	Traditional BMS	Reduced effort BMS
	• Distribution of the samples over the region (sampling design): Preferably random/systematic sampling (e.g. as in France or with wider-countryside BMS in UK). But the number of volunteers willing to participate in counting sometimes unattractive sites might limit the possibilities for random or systematic sampling. • Time frame during the season: weekly or two-weekly counts. • Fully tested, success proven.	• Distribution of the samples over the region (sampling design): Preferably random/systematic sampling (e.g. as in France or with wider-countryside BMS in UK). • Time frame during the season: UK: three visits within nine weeks with a one week gap. F: four visits in four months, with 15 days in between. • Some full traditional BMS sites will likely be needed in a reduced effort scheme - to calibrate data and help analyse the results. • The reduced effort BMS is work ongoing and has not been fully tested.

Butterfly Monitoring Methodology

Field methods

All schemes apply the method developed for the British Butterfly Monitoring Scheme (Pollard and Yates 1993). The counts are conducted along fixed transects of about 1 kilometre, consisting of smaller sections, each with a homogeneous habitat type. The fieldworkers record all butterflies 2.5 metres to their right, 2.5 metres to their left, 5 metres ahead of them and 5 metres above them (Van Swaay et al. 2002). Butterfly counts are conducted between March-April to September-October. Visits are only conducted when weather conditions meet specified criteria. In the Dutch (and German) scheme this means temperature above 17°C, or 13–17 °C in sunny weather, windspeed less than 6 on the scale of Beaufort and no rain (Van Swaay et al. 2002). Most of the transects are recorded by skilled volunteers, but their results are usually checked by butterfly experts.
The number of visits varies from every week in the UK and the Netherlands to 3-5 visits annually in France. In the Netherlands, transects dedicated to rare species can be visited only during the expected flight period of the species. In normal transects, weekly counts cover the entire flight period of species and thereby offer the opportunity for assessing temporal population trends per transect, but the precision of the trend estimates may be limited (Harker and Shreeve 2008). Weekly visits may however also be demanding for observers. If the objective is only to produce large scale (e.g. national) trends, the efforts may be reduced to much fewer visits (Heliölä and Kuussaari 2005; Roy et al. 2007). Such a reduced-effort scheme is planned in the UK for the wider countryside where mainly common butterflies occur and few volunteers can be recruited. This proposed reduced-effort scheme is based on only a few annual visits, targeted to the period when most information can be gathered, i.e. three visits in July–August plus in some cases an additional one in May (Roy et al. 2005; 2007). Yet a problem with the reduced effort schemes can be that it will often not be possible to compare different regions, habitats or management regimes to find the underlying drivers for population changes. Furthermore much more transects will be needed in a reduced effort scheme than in a traditional scheme. The main characteristics of the 'Traditional' and 'Reduced effort' schemes are summarized in Table 5.2.
Observers never detect all butterfly individuals present during their visit in the study area (Dennis et al. 2006; Kéry and Plattner 2007). Therefore, transect counts do not provide information on absolute butterfly numbers but rather yield species-specific relative abundance indices that are assumed to reflect year-to-year

population changes over the entire study area. The assumption of constant detection probability has been underpinned by the demonstration of close correlations between transect counts and population estimates based on mark-recapture data (Pollard 1977; Thomas 1983). However, if for some reasons the detection probability for a given species varies over time then trends inferred from transect count results uncorrected for this probability may be biased (Kéry and Plattner 2007).
The likely sources of between-year variation in detection probability are e.g. weather, time of day, observer experience, and vegetation height changing due to succession or more generally any habitat changes (Pollard et al. 1986; Harker and Shreeve 2008; Pellet 2008). Variation due to weather and time of day can be reduced by standardisation of the conditions in which transect counts are conducted (Pollard 1977; Pollard et al. 1986). In addition, in the case of large-scale and long-term monitoring such variation may be assumed to be random only, thereby decreasing the precision of the results, without inducing any bias. Still, any systematic changes in observer experience, vegetation height or even the behaviour of species cannot be ruled out completely. We therefore suggest to test any long-term changes in detection probabilities using capture-recapture methods as applied for butterflies by Kery and Plattner (2007) and Pellet (2008) or distance-sampling methods (Pollock et al. 2002). Distance sampling has already been applied in butterfly population studies in Northern America (Brown and Boyce 1998), and there are currently attempts to incorporate it in the UK Butterfly Monitoring Scheme (K. Cruickshanks, pers. comm.).
A related problem is that of the variable longevity in adult butterflies and its effect on transect count reliability. Since adult butterflies typically eclose in daily cohorts, their numbers recorded on transects depend not only on seasonal population sizes, but also on longevities, and consequently transect count results do not necessarily follow year-to-year population changes precisely (Zonneveld 1991; Nowicki et al. 2005; 2008). Nevertheless, the effect of between-season variation in butterfly longevity is likely to become random with extensive data sets.

Transect selection

To be able to draw proper inferences on the temporal population trends at national or regional level, transects should best be selected in a random or stratified random manner (Sutherland 2006). Several recent schemes, e.g. in Switzerland and France, have been designed in this manner (Henry et al. 2005). Unfortunately, such a procedure would yield many data for common butterflies, but few data for rare butterflies, unless an unrealistically high number of transects is selected. If a scheme aims to monitor rare species, scheme coordinators preferably locate transects in areas where rare species occur, leading to an overrepresentation of special protected areas. In the older schemes, such as in the UK and the Netherlands, but also in the recently established scheme in Germany, transects were selected by free choice of observers, which in some cases has led to the overrepresentation of protected sites in natural areas and the undersampling of the wider countryside and urban areas (Pollard and Yates 1993); while in Germany this effect was not that pronounced (Kühn et al. 2008). Obviously, in such a case the trends detected may be only representative for the areas sampled, while their extrapolation to national trends may produce biased results. Such bias can however be minimized by post-stratification of transects. This implies an a posteriori division of transects by e.g. habitat type, protection status and region, where counts per transect are weighted according to their stratum (Van Swaay et al. 2002, see also Henry et al. (2008) for the principles of weighting).

Calculating indices and population trends

The traditional way of testing temporal population trends in yearly count data is to apply ordinary linear regression. But linear regression assumes the data to be normally distributed, which does not hold for most count data especially if the data contain many zero values. Also log transformation does not work properly in such cases. Generalized Linear Models (GLM; McCullagh and Nelder 1989) offer an alternative to analyse count data. In GLM models, the normality assumption is replaced by the assumption of a distribution of the user's choice. For count data this distribution is often the Poisson distribution. To apply these models transformation of raw data is no longer required. Poisson (or loglinear) regression is implemented in the widely used program TRIM (TRends and Indices for Monitoring data - Pannekoek and van Strien 2005). Regarding butterflies, this program is used in the UK and the Netherlands and new schemes plan to use it as well (Kühn et al. 2008). Based on a model with year effects and site effects, TRIM produces yearly indices as well as overall trend estimates and is particularly useful if the data contain missing counts due to the coming and going of the voluntary observers in a scheme. TRIM has also options to incorporate serial correlation between counts in consecutive years, testing of covariates and testing of changepoints. An important feature of TRIM is the possibility to incorporate weight factors per transect in order to adjust for oversampling and undersampling of particular habitat types, regions or other characteristics of transects. These weights may be based on e.g. the surface area of heathland in different regions for heath butterflies, or the population shares of species per region (Van Swaay et al. 2002). One might also consider to apply detection probabilities as weights in TRIM, if these probabilities appear to change over time.

A weakness of TRIM is that the model does not include week effects. The counts per week need to be combined first into a yearly sum and only this sum enters the TRIM model. Rothery and Roy (2001) explored the possibilities to apply Generalized Additive Models (GAM) to butterfly monitoring data. A GAM is an extension of GLM methods and allows the smoothing of yearly indices.

Applications

National and regional trends

The main objective of most butterfly monitoring schemes is the production of regional and/or national population trends. These trends are being produced on a routine basis every year in e.g. the UK and the Netherlands, and are meant to evaluate at a large scale the need for or the progress made in butterfly conservation.

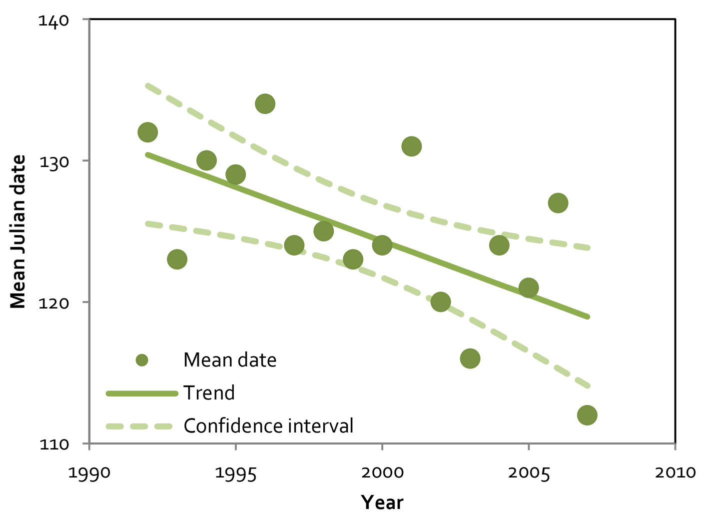

Figure 5.2: Mean Julian date of the first 10% of all observed individuals of 19 spring butterfly species in 1992-2004 (January 1 = day 1 etc.). For each species the date was assessed per year of the first 10% of all observed individuals in the entire flight period on all transects together. For details see Van Strien et al. (2008). Trends and confidence intervals were assessed by structural time-series analysis and the Kalman Filter using the program Trendspotter (Soldaat et al. 2007).

Relationships with environmental factors

The transect counts can be used to study the relationships with environmental factors, such as climate change, nutrient load, heavy metals, drainage, land use, fragmentation and management practice. Pollard and Yates (1993) describe detailed studies based on monitoring data. Here we mention only a few examples:

- Climate change. Several schemes were used to examine the changes in phenology (Roy and Sparks 2000; Stefanescu et al. 2003; Kühn et al. 2008; Van Strien et al. 2008, see figure 5.2). WallisDeVries and Van Swaay (2006) used transect data to study the effects of the combination of nitrogen deposition and climate change on the abundance of butterflies.
- Nutrient load and heavy metals. Oostermeijer and Van Swaay (1998) examined relationships between butterfly absence/presence data obtained from monitoring transect and Ellenberg indicator values for nutrients, acidity and moisture (figure 5.3). Mulder et al. (2005) examined the effects of heavy metals on butterflies on a particular transect.
- Management practice. Brereton and Warren (2005) found the trend of *Lysandra coridon* on calcareous grasslands with butterfly friendly management to be more positive than on other grasslands.
- Multiple environmental factors. Other perspectives for the application of monitoring data are by testing predictions or expectations from envelope approaches, which form the basis of many biodiversity impact and risk assessments (as e.g. in the ALARM project; Settele et al. 2005). This may in particular be relevant to large scale predictions/expectation of changes and trends derived from the combined effects of a multitude of pressures (compare Schweiger et al., in rev.) and to extrapolations of historically reconstructed trends (Settele et al. 1992).

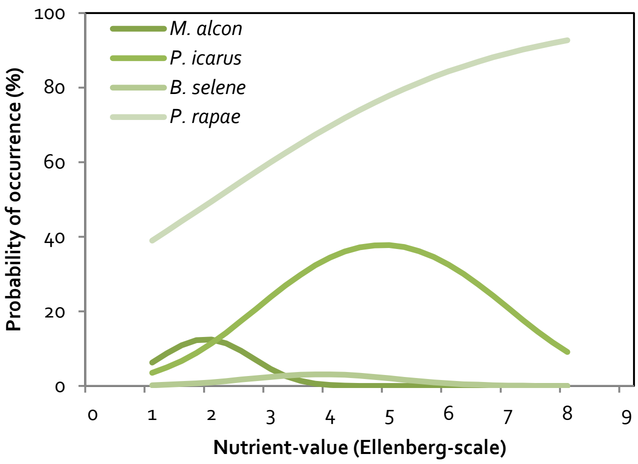

Figure 5.3: Relationships between the probability of occurrence obtained from monitoring transect data and Ellenberg indicator values for nutrients (from Oostermeijer and Van Swaay 1998).

Butterflies as indicators

Government representatives at the 2002 World Summit of Sustainable Development pledged 'a significant reduction in the current rate of biodiversity loss by 2010'. The commitment of the EU to protecting biodiversity is even stronger by aiming at halting biodiversity loss by 2010 (Balmford et al. 2005; Gregory et al. 2005). Butterflies may be useful as biodiversity indicators for reporting on the development towards the EU 2010 target. Contrary to most other groups of insects, butterflies have considerable resonance with both the general public as decision-makers (Kühn et al. 2008). Butterflies are also relatively easy to recognize and data on butterflies has been collected for a long time and by many voluntary observers. The method is well described, extensively tested and scientifically sound (Pollard 1977; Pollard and Yates, 1993) As a result butterflies are the only invertebrate taxon for which it is currently possible to estimate rates of decline among terrestrial insects in many parts of the world (de Heer et al. 2005; Thomas 2005). However, butterflies can only be regarded as good biodiversity indicators if it is possible to generalise their trends to a broader set of species groups (Gregory et al. 2005). Admittedly, there is currently a heated debate on how well butterflies meet this criterion. Hambler and Speight (1996; 2004) claimed that this group is likely to experience greater declines than other organisms due to their herbivorous life strategies and thermophily, but Thomas and Clarke (2004) convincingly rejected both arguments. Based on a comprehensive review of studies into their life-history traits, relative sensitivity to climate change, and adjusted extinction rates Thomas (2005) concluded that butterflies may be considered representative indicators of trends observed in most other terrestrial insects, which together form a major fraction of biodiversity.

Trends per butterfly species can be combined into a unified measure of biodiversity. We followed Gregory et al. (2005) in averaging indices of species rather than abundances in order to give each species an equal weight in the resulting indicators. When positive and negative changes of indices are in balance, then we would expect their mean to remain stable. If more species decline than increase, the mean should go down and vice versa. Thus, the index mean is considered a measure of biodiversity change. We used geometric means rather than arithmetic means, because we consider an index change from 100 to 200 equivalent, but opposite, to a decrease from 100 to 50. Buckland et al. (2005) discussed a number of possible composite indicators and found the geometric mean of indices a useful approach.

The results of national butterfly monitoring schemes may be combined to create an indicator at a supra-national level (see also Henry et al., 2008). Based on the procedure described for European birds (see Gregory et al., 2005), a preliminary grassland butterfly indicator has been developed (Van Swaay and Van Strien 2005). The procedure was as follows:

1. National level. The indices for each species were produced for each individual country with a butterfly monitoring scheme, using TRIM (Pannekoek and Van Strien 2005). Figure 5.4 shows the national indices as an example for three countries for the grassland species *Lasiommata megera*.

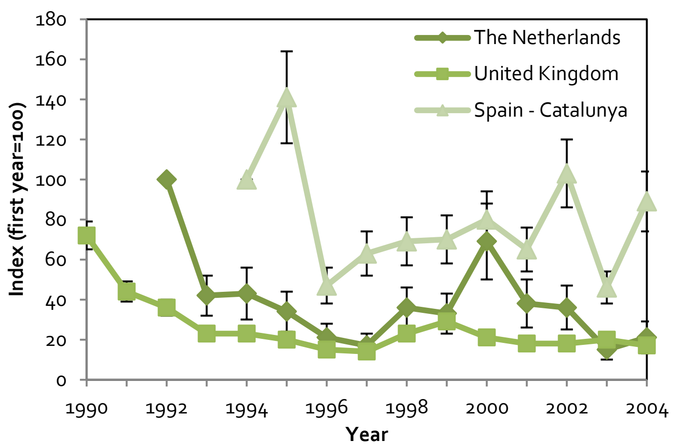

Figure 5.4: National abundance indices (± standard error) for **Lasiommata megera** ***in three European countries. In the first year the index can be calculated it is set to 100 (1992 for The Netherlands, 1994 for Catalunya, 1976 for the United Kingdom).***

2. Supranational level. To generate supra-national trends, the difference in national population size of each species in each country was taken into account. This weighting allows for the fact that different countries hold different proportions of a species' European population (Gregory et al., 2005). Here, we applied as weights the proportions of each country (or part of the country) in the European distribution of a species (based on Van Swaay and Warren 1999). The missing year totals are estimated by TRIM in a way equivalent to imputing missing counts for particular transects within countries (Gregory et al. 2005). Figure 5.5 gives the weighted and combined trend for *Lasiommata megera*. The same procedure may be used to establish European trends for the Habitats Directive species e.g. *Euphydrias aurinia, Maculinea arion and M. nausithous* (which are included in the grassland indicator).
3. Multispecies level. For each year the geometric mean of the supranational indices is calculated. The preliminary grassland indicator was based on seven widespread grassland species (*Ochlodes venata, Anthocharis cardamines, Lycaena phlaeas, Polyommatus icarus, Lasiommata megera, Coenonympha pamphilus, Maniola jurtina) and ten grassland-specialists (Erynnis tages, Thymelicus acteon, Spialia sertorius, Cupido minimus, Maculinea arion, Maculinea nausithous, Polyommatus bellargus, Polyommatus semiargus, Polyommatus coridon, Euphydryas aurinia*).

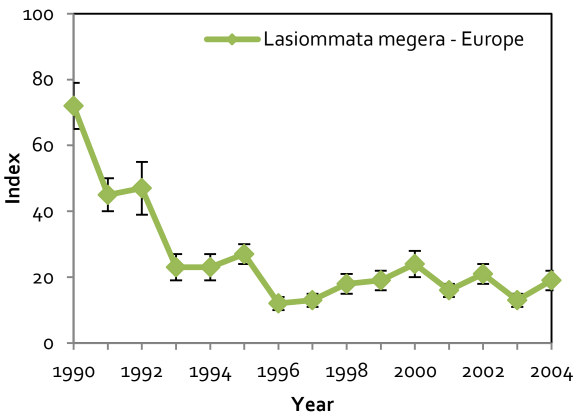

Figure 5.5: Collated index (± standard error) for* Lasiommata megera *in the European countries with Butterfly Monitoring Schemes.

The countries covered were mainly from Western Europe (Table 5.1). The average grassland butterfly abundance appeared to decline by almost 50% (Figure 5.6), which is most probably linked with the agricultural intensification in Western Europe (Van Swaay and Warren 1999; Gregory et al. 2005). The decline is much stronger than the decline of the farmland bird indicator, which has fallen by 19% in the same period (Gregory et al. 2008). This corresponds with the findings in the UK where butterflies have experienced greater losses than birds (Thomas et al. 2004).

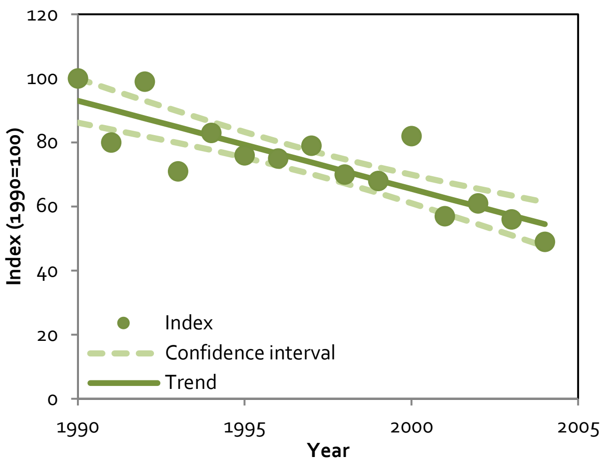

Figure 5.6: European grassland butterfly indicator. Trends and confidence intervals were assessed by structural time-series analysis and the Kalman Filter using the program Trendspotter (Soldaat et al. 2007).

Perspectives

The number of countries with butterfly monitoring schemes is increasing. In addition, the quality of schemes is improving, because any lack of representativeness of transect is taken into account, either by choosing an adequate design or to adjust any bias during the stage of analysis. As the number and quality of butterfly monitoring schemes grows, the coverage of Europe by supranational species trends and multispecies indicators improves. The European Environmental Agency has already recommended to develop European butterfly indicators (European Environment Agency 2007), and these developments may lead to indicators that are comparable to the farmland bird indicator, which has been adopted by the EU as biodiversity indicator (Gregory et al. 2005). Where possible and feasible, one might even think of combining butterflies and birds in indicators to report against EU's 2010 target, in order to generalize changes well beyond the set of species.

The grassland butterfly indicator offers the possibility to detect large scale effects of either abandonment of agricultural land (especially occurring in Eastern and Southern Europe) or intensification of agricultural practices (a process already stopped in parts of Western Europe, but ongoing in many European regions). Apart from a grassland butterfly indicator, we see good perspectives to create a climate change indicator, summarising changes in occurrence of species driven by climate change, as well as a woodland indicator. The same indicators are also in progress for European birds (Gregory et al., 2007). A woodland indicator may however not have such a simple message as the preliminary grassland indicator. That is because woodland butterflies are made up of two different species groups. The first group of woodland butterflies are characteristic for woodland edges and open spots, e.g. *Euphydryas maturna* and *Coenonympha hero*. The second group are canopy species, who profit from high forest, e.g. *Apatura iris*. Though both these groups are genuine woodland butterflies, their expected trends differ entirely. Species from the first group probably suffer in large parts of Europe, because traditional coppicing has been replaced by management for high forest. In Western Europe, where this process has been going on for a few decades, these species are virtually extinct, but in Eastern Europe strong populations still exist (Van Swaay and Warren 1999; 2003). The few species of the second group, which tolerate dense forests (e.g. *Pararge aegeria*; Shreeve 1984) or the handful of European canopy dwellers (e.g. *Neozephyrus quercus*, *Apatura* spp., or *Limenitis populi*) are rather the exception. Thus, a woodland indicator might have to consider a differentiation of these two groups. As a rule, the majority of European woodland butterflies utilises sunny habitats within woodlands, such as sparse stands, bogs, streamsides, clearings, rides, or edges (Settele et al., 2008).

Over thirty years butterfly monitoring has developed from one test site in Monks Wood in the United Kingdom to more than 2000 transects scattered over Europe. Almost every year new countries join in to start up a monitoring network. Further extension of butterfly monitoring schemes to other countries in Europe should be encouraged and supported by the European Union and its Member States. The further development and use of butterflies in European biodiversity indicator will further stimulate new butterfly monitoring schemes.

Acknowledgements

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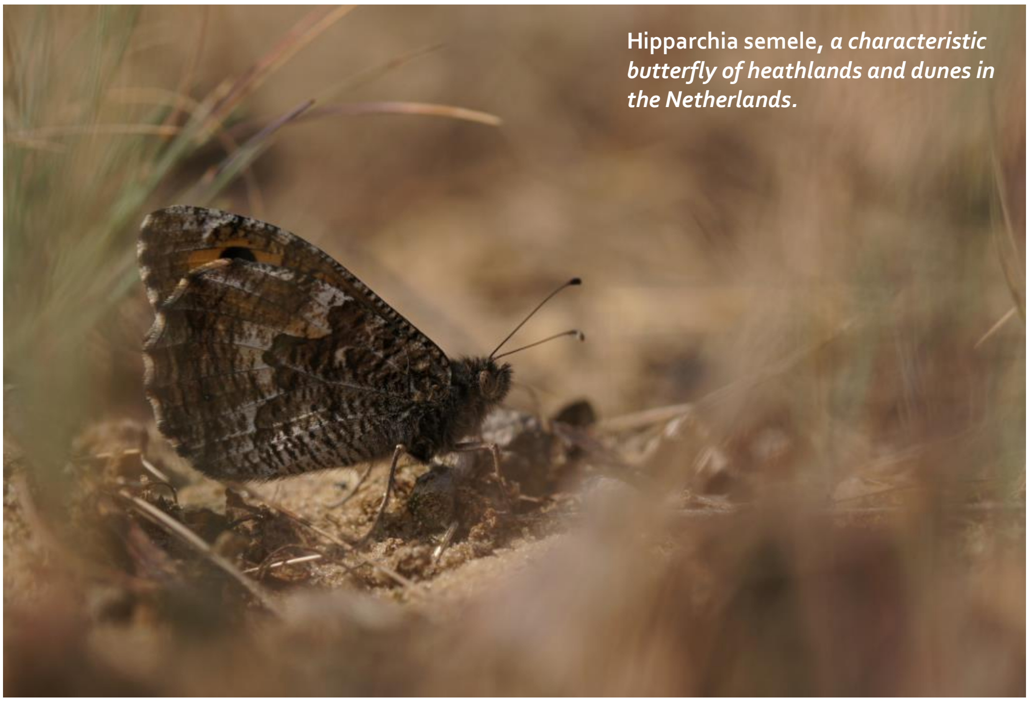
6. Monitoring butterflies in the Netherlands: how to get unbiased indices

*Slightly modified from: Van Swaay, C.A.M., Plate, C.L. & Van Strien, A.J. (2002) Proc. Exper. Appl. Entomol. NEV Amsterdam **13**, 21–27.*

Abstract

The Dutch Butterfly Monitoring Scheme started in 1990. In 2002 more than 300 sites are monitored yearly, most of them by volunteers. The main results are national yearly indices per species describing changes in species abundance. Since the monitoring sites are not equally distributed over the country, oversampling and undersampling of particular regions and habitat types may lead to biased estimates of the national indices. In this paper we present a method to adjust for unequal sampling using *Hipparchia semele*, a characteristic species of heathlands and dunes as an example.



Hipparchia semele, *a characteristic butterfly of heathlands and dunes in the Netherlands.*

Introduction

In the last century, many butterflies in the Netherlands have declined in range and abundance. Of the 70 native Dutch butterfly species, 17 have become extinct and 30 are considered threatened on the red list (Van Ommering et al., 1995; Maes & Van Swaay, 1997). Only 23 species are considered as 'safe and/or low risk'.

In 1990, De Vlinderstichting (Dutch Butterfly Conservation) and CBS (Statistics Netherlands) started a Butterfly Monitoring Scheme in the Netherlands (Van Swaay et al., 1997; Van Swaay, 2000a). The main objective of the monitoring scheme is to assess changes at national and regional level of common and rare butterfly species, including species of the Habitat Directive. These changes will provide useful information on the success of nature conservation policy tools, like red lists (e.g. RIVM et al., 2001; CBS et al., 2001). In addition, the monitoring data can be applied to the conservation of butterflies at a local level by comparing national changes with local changes (e.g. Van Swaay, 2000a). Finally, the data are also useful for research purposes, for instance to examine and evaluate the effects of specific conservation measures on butterflies (e.g. Wallis de Vries & Raemakers, 2001).

Because it is not possible to count all individual butterflies to establish the true trend in species abundance, it is necessary to take samples. Because volunteers form the major part of the recorders, each of them with particular preferences, the sites are not equally distributed over the Dutch landscapes and habitat types. This has the risk of biased estimates of the national changes. Here we describe a method to adjust for this unequal sampling.

Method

Fieldwork

To a large extent the field method is based on the British Butterfly Monitoring Scheme (Pollard & Yates, 1993). Only a few minor changes have been made. The most important adaptation is that all transects have been divided into sections with a fixed length of 50 m. Such a section must have a homogeneous vegetation structure. The length of a transect can be up to 1 km (20 sections), but may be shorter. From April to September all butterflies 2.5 m to the left and right of the recorder and 5 m in front and above should be counted weekly under standardized weather conditions. The method is described in detail in Van Swaay (2000b). Most of the sites are recorded by volunteers.

This method has proved to be successful in collecting a large set of data for common and widespread species. For rare species, however, it was not possible to get data from a sufficient number of sites. Therefore, since 1994 'single species sites' have been added. At these sites only one species is counted in its flight period. This reduces the number of required visits to those sites from twenty to about four, thereby increasing the opportunities of nature reserve wardens and volunteers to count these species sufficiently.

Calculating the year-count per site

At the end of the season all recorders send in their data on standard paper forms. After a first check by butterfly specialists of Dutch Butterfly Conservation, Statistics Netherlands (CBS) performs standardized checks by computer programmes to detect typing errors and other errors. Thereafter recorders are asked to check these errors.

Over the flight period of a particular butterfly species, a series of counts is obtained for each transect (see example in figure 6.1). The number of butterfly individuals rises and goes down during the flight season, due to the emergence of butterflies from pupae or by immigration, followed by death or emigration.

Transects that are not counted sufficiently often during the season should be disregarded for further use, because the peak of the numbers may be missed entirely. We have applied the following procedure to select the transects that are counted sufficiently. At first, for each particular year, we have assessed the flight period of each generation of all species. This was derived from the mean number of butterflies per week across all sites. For instance, the flight period for *Hipparchia semele* in 1999 was between week 27 and 36 (figure 6.2). If there is an overlap between the second and third generation, which happens in the Netherlands almost every year for the *Pieris* species, these two generations were taken together. For species that overwinter as adults, the butterflies emerging in summer resulting from the eggs laid in spring, are regarded as the first generation. Thereafter, we selected only those transects for further use that (i) were at least counted once in the middle of the flight period and (ii) on which the time between two subsequent visits was not longer than half of the flight period. Transects that did not fulfil these criteria were omitted and a missing value was added for that particular year. A transect that does not fulfil the criteria for one species, may very well be used for other species. Finally, we assessed the total year-count for each site for each particular year and species. This is an estimate of the area under the line (the number 'butterfly days') connecting the individual counts for each species per transect per year (see figure 6.3). At the start and end of the flight period we assumed to have a zero count. Because transects are counted once a week, this area is divided by seven. It is calculated as:

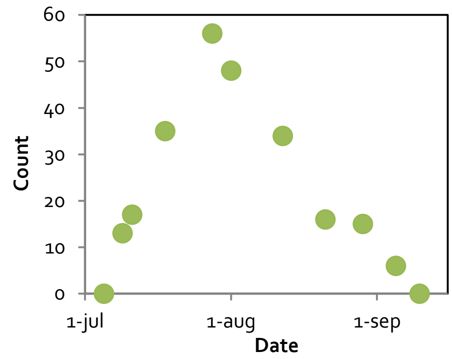


Figure 6.1: Example of the data in the Dutch Butterfly Monitoring Scheme: individual counts of Hipparchia semele *in 1999 on a transect in Berkheide (dune-area near Leiden, province of Zuid-Holland).*

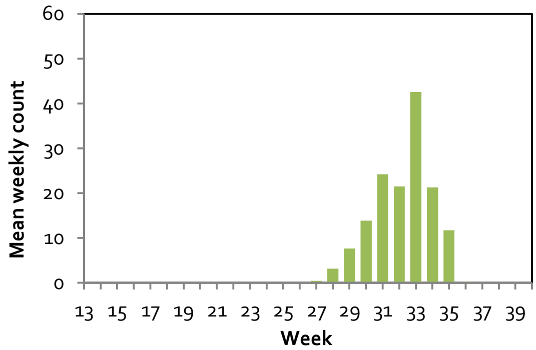


Figure 6.2: Mean count in each recording week for Hipparchia semele *in The Netherlands in 1999.*

$$J = \sum_{i=a}^{i=b-1} \frac{1}{2}(t_{i+1} - t_i)(N_i + N_{i+1})/7$$

where J = year-count, i = number of visit, a = first visit, b = last visit, t = day number, $t_{i+1} - t_i$ = length of period between 2 visits in days, N = count at visit

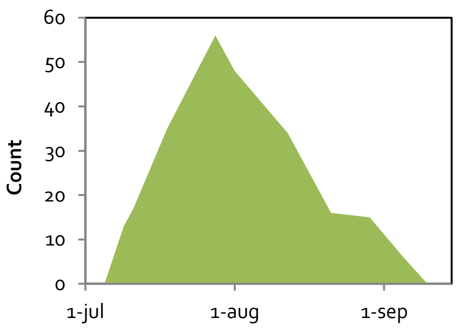


Figure 6.3: For the year count the area under the line connecting the individual counts of Hipparchia semele *per transect per year (see figure 1) is calculated.*

Calculating trends and indices across sites

The changes of species are presented as indices, using the first year as a base year. To be able to calculate reliable indices a minimal number of transects per stratum is needed. In the first two years of the Dutch Butterfly Monitoring Scheme the number of transects was relatively poor for many strata. For this reason the first year for the results of the weighted analysis is 1992.

Indices were calculated using the computer program TRIM (Pannekoek & Van Strien, 2001). TRIM is an index program for the analysis of time series of counts with missing data, based on loglinear regression. Monitoring data often contain many missing values. The idea is to estimate a model using the observed counts and then to use this model to predict the missing counts. Indices can then be calculated on the basis of a completed data set with the predicted counts replacing the missing counts. For species with more than one generation per year the index of the first generation has been used for greater accuracy according to Van Strien et al. (1997). Due to lack of data for the first generation for *Aricia agestis* and *Lycaena tityrus* the second generation has been used. In addition to indices, overall trends across the entire period were calculated using TRIM.

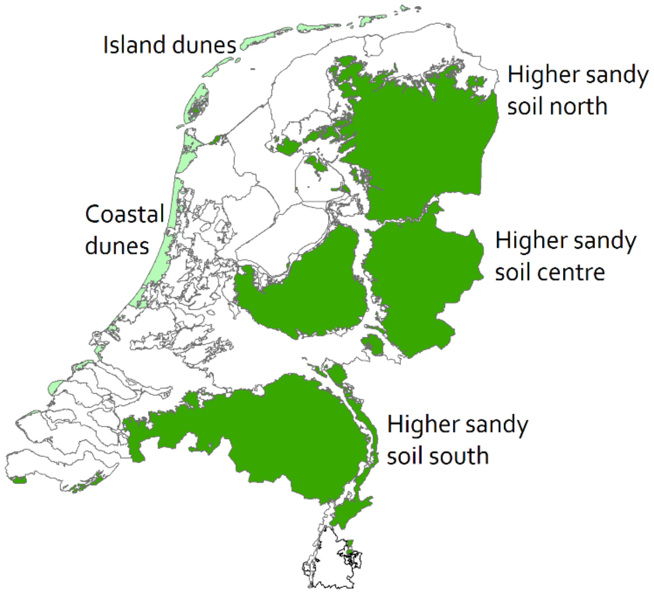


Figure 6.4: The Dutch physical geographical regions.

Weighted trend analysis

The national indices are calculated by using a weighting procedure. This is necessary because butterflies and transects are not equally distributed over the country and the habitats of butterflies. In order to counter for this uneven sampling, we have applied a post-stratification of the data and have calculated indices and trends for each stratum separately. A stratum consists of a combination of (i) a Dutch physical geographical region, such as the Northern higher sandy soil area (figure 6.4) and (ii) one of the following habitat types: woodland, heathland, agricultural land, moorland, dunes and urban areas. Thereafter, we have added the strata weighted together to get a more correct estimation of the national indices. If all strata are equally sampled according to their surface area, all weight would be similar. If a stratum is undersampled, it should get a higher weight than other strata. This weight should be higher when the stratum is more important for the species. Thus, the weight factors are based on the distribution of the transects across the strata and the relevance of each stratum for each particular species. In order to assess this relevance, first the surface area of each the stratum has been calculated using a GIS with a map of habitat types, whereby we took into account only the area in which the species occurs according to its distribution in the Netherlands. In addition, we estimated the relevant part of each habitat type per species by expert judgement. This is necessary because a habitat area is not entirely appropriate for the species. For example, many woodland butterflies are restricted to the edges and do not occur in the dark forest interior. The weighting factor for a stratum is the quotient of the number of transects per stratum in case all transects are distributed proportionally to their relevance and the actual number of transects per stratum.

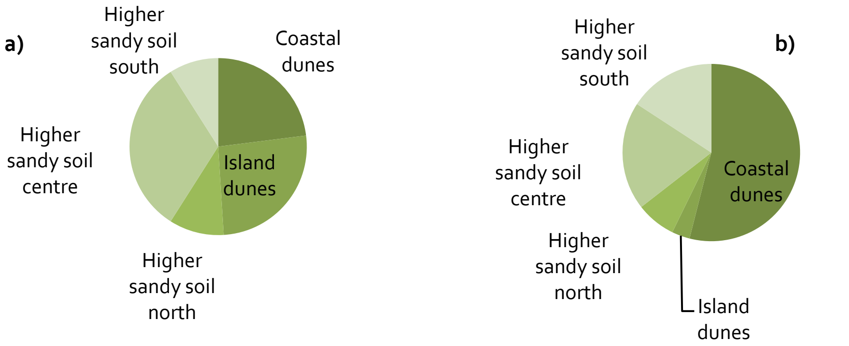
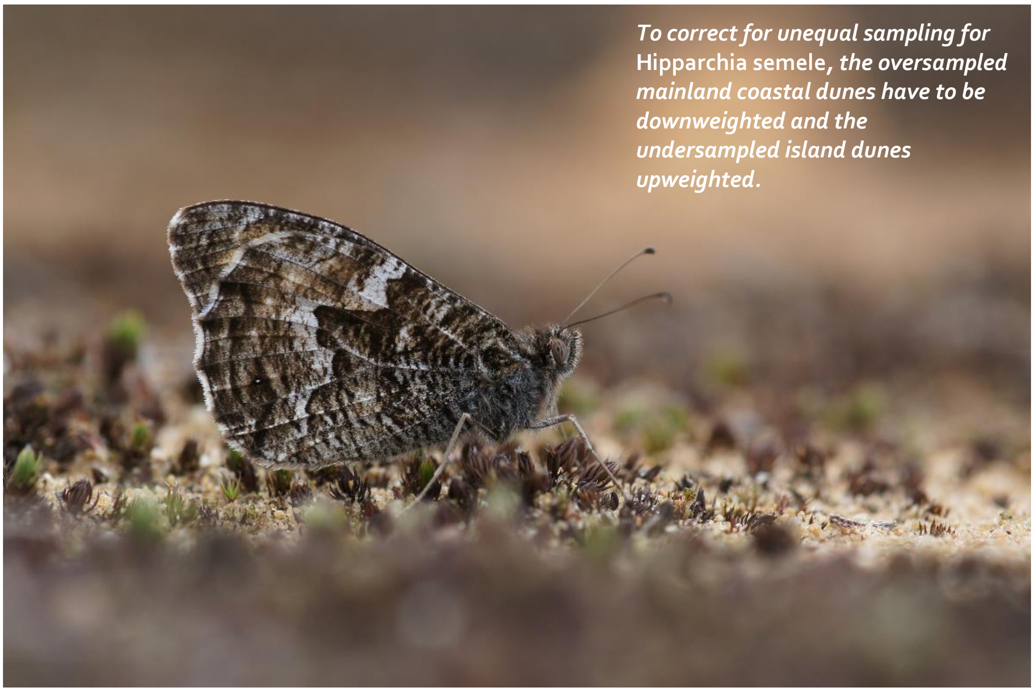


Figure 6.5: Distribution of a) the estimated habitat area of* Hipparchia semele *over the five strata where it occurs and b) of the transects where the butterfly is recorded over the different strata (n=54 transect in 2002).

The procedure can be illustrated by *Hipparchia semele*, a characteristic butterfly of heathlands and dunes. Figure 6.5a gives the distribution of the relevant area of *Hipparchia semele* over the five strata where it occurs. Figure 6.5b shows the distribution of the transects where the butterfly is recorded over the different strata. It is clear that the dunes on the mainland are oversampled ($\chi 2=108.9$, $p<0.001$). To adjust for this bias in the case of *Hipparchia semele* the oversampled mainland coastal dunes are down weighted by 0.43, for the undersampled island dunes on the Wadden islands the factor is 8.0.



Results

For 37 butterfly species weighted trends have been calculated (table 6.1). Six species show an increase, but for *Coenonympha pamphilus* it should be noted that the numbers of this species showed a massive decline in 1990-1991. The strong increase since 1992 has not yet compensated for the loss in the earlier years. The only red list species showing an increase is *Papilio machaon*, benefiting from increasing summer temperatures in the last decade.
Six species are more or less stable, but fifteen butterflies show a decrease in numbers after 1992. Most of them are red list species, like Hipparchia semele, but also widespread and abundant butterflies like *Aglais urticae* and *Gonepteryx rhamni* have declined.
The trend of four species remains unknown due to high standard errors caused by strong fluctuations in time or very different trends between transects.
The weighted results for *Hipparchia semele* show a stronger decline as compared to the unweighted results (figure 6.6). The cause of this difference can be seen by looking at the indices for each separate stratum (figure 6.7).

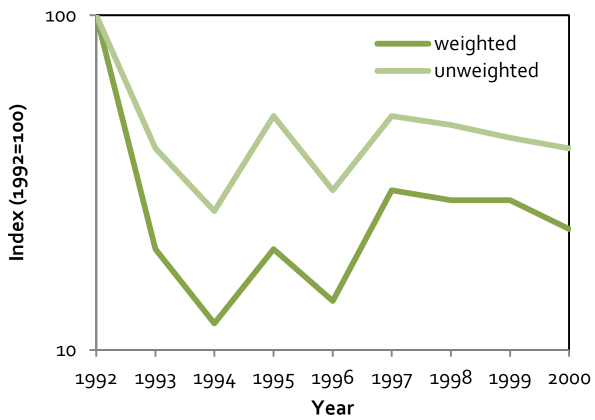


Figure 6.6: National weighted and unweighted indices of* Hipparchia semele *in The Netherlands. A weighted index is corrected for the unequal distribution of transects over the range of the species, with some areas over- and others undersampled.

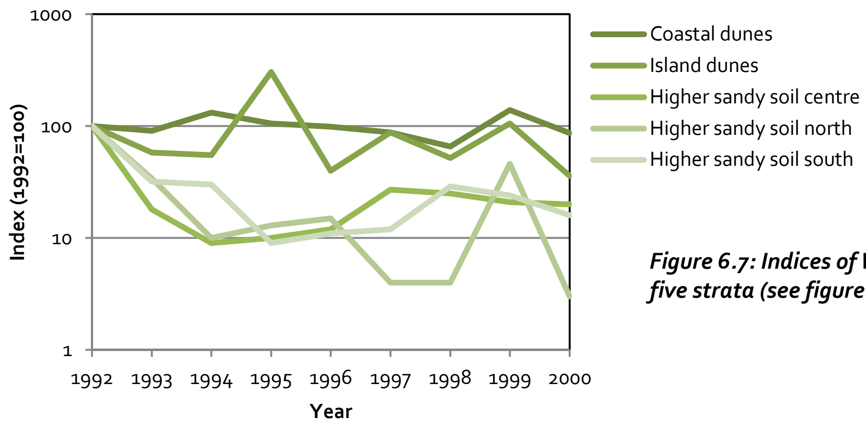


Figure 6.7: Indices of* Hipparchia semele *in five strata (see figure 6.4).

Table 6.1: Evaluation of the weighted national trends of the Dutch butterflies from 1992-2000. For species with more than one generation a year the trend for the first generation is given, except for Plebeius agestis *and* Lycaena tityrus *where the second generation is used.*

	Species	Evaluation	Magnitude
Increase: 6 species	*Pararge aegeria*	Significant very strong increase	>75% in 5 years
	Lycaena phlaeas		
	Coenonympha pamphilus	Significant strong increase	50-75% in 5 years
	Pieris napi	Significant moderate increase	25-50% in 5 years
	Celastrina argiolus	Probable increase	
	Papilio machaon		
Stable: 12 species	*Lasiommatua megera*	Stable	<25% change in 5 years
	Polygonia c-album		
	Pieris brassicae		
	Araschnia levana		
	Anthocharis cardamines		
	Aricia agestis (2nd generation)	More or less stable	<50% change in 5 years
	Pyrgus malvae		
	Carterocephalus palaemon		
	Erynnis tages		
	Polyommatus icarus		
	Pieris rapae		
	Aphantopus hyperantus		
Decrease: 15 species	*Maniola jurtina*	Significant small decrease	<25% in 5 years
	Aglais io	Significant moderate decrease	25-50% in 5 years
	Hipparchia semele		
	Favonius quercus		
	Thymelicus sylvestris		
	Callophrys rubi		
	Plebejus argus		
	Issoria lathonia		
	Pyronia tithonus		
	Thymelicus lineola		
	Satyrium ilicis	Significant strong decrease	50-75% in 5 years
	Gonepteryx rhamni		
	Limenitis camilla		
	Aglais urticae		
	Hesperia comma		
	Melitaea athalia	Significant very strong decrease	>75% in 5 years
	Heteropterus morpheus		
Unknown: 4 species	*Lycaena tityrus* (2nd generation)	Strong fluctuations	
	Ochlodes sylvanus		
	Argynnis aglaja		
	Boloria selene		

Discussion

In an ideal monitoring scheme:

- plots are randomly selected (stratified) from the species distribution areas;
- all observers participate from the beginning;
- all observers count every week;
- all observers are equally experienced;
- an observer never ends its participation.

But because we do not live in an ideal world, we have to face a number of problems that may distort the results. Here we have dealt with probably one of the most important problems, the unequal sampling. In the Netherlands this is especially the case in the coastal dunes. Many recorders prefer to count butterflies in this relatively unspoilt area over the agricultural and urban areas. On the other hand, the dunes of the Wadden islands are undersampled. Although these islands attract many butterfly enthusiasts for short holidays, there are only a few local people who do a monitoring transect. For some species this leads to biased indices if all transects are calculated without correcting for this phenomenon.
Both Coastal and Islands areas show a more or less stable trend for the dunes. But this species has declined severely on the heathlands, especially in the Northern part of the country. The index for 2000 here is less than 10% of the base year 1992, which is an enormous decrease! Because the mainland dunes are heavily oversampled (figure 6.5), the unweighted national results are biased towards a more stable trend. As a result of weighting, the overall indices present a more realistic view on the development of this butterfly species in the Netherlands.
A major assumption of the procedure applied is that transects are representative for each stratum. But this might not be true. A recorder may start a transect in order to evaluate the development of butterfly numbers to special nature management actions or a recorder may lose his motivation for the monitoring when the numbers of butterflies have become low. Such suspected reasons to start and stop the counting should be taken into account. Furthermore, even within a specific habitat, recorders may have a strong preference to count butterflies in the most attractive parts. The trends of butterflies in these parts might differ from the trends in the rest of the stratum. Further research of these phenomena is required to find out how serious these problems are and how they can be solved.

Acknowledgements

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7. Developing a butterfly indicator to assess changes in Europe's biodiversity

*Slightly modified from: Brereton, T.; Swaay, C.A.M. van & Strien, A.J. van (2009) Avocetta **33**: 19-27*

Abstract

To monitor progress towards the European Union target to halt the loss of biodiversity by 2010, biodiversity indicators at a European scale are required. Butterflies have been proposed as biodiversity indicators due to their rapid and sensitive responses to subtle habitat and climatic changes and as representatives for the diversity and responses of other wildlife, especially insects. Since the first butterfly monitoring scheme was established in the UK in 1976, schemes have now been established in over ten European countries. In each scheme, regular butterfly counts are made through the season each year along fixed routes under suitable weather criteria. Here, we used the counts to compile both national and supra-national annual indices for a number of species, in order to develop and test a preliminary European scale biodiversity indicator for the European Environment Agency. A multi-species grassland "European" Butterfly indicator was constructed by combining data from 17 characteristic grassland species, following closely the analytical method developed for the European Bird Indicator. The indicator showed a strong decline in butterfly abundance (of about 40% in 15 years since 1990). The European Environment Agency has subsequently proposed a number of indicators for inclusion in the set of European biodiversity indicators, butterflies being one of the most highest ranked. We hope to update and develop the indicator further (including compiling an indicator for woodland butterflies), make further analytical improvements and extend butterfly monitoring schemes to other countries in order to improve the quality and representativeness of the indicator.

Abandonment leads to grasslands overgrown by scrub and trees leaving no habitat for grassland butterflies.

Introduction

Recent years have seen global political consensus on the need to address the loss of biodiversity. The 1994 Convention on Biological Diversity (CBD) put an obligation on individual governments to develop national strategies for the conservation and sustainable use of biological diversity. As part of the response, in 2001 the European Union set an ambitious target to halt biodiversity loss across Member States by 2010, which was backed up by agreement under international law in 2002 through the CBD. In 2006, the EU published an Action Plan as a road map to delivering the 2010 target, including concrete measures and outlining the responsibilities of EU institutions and Member States. An important component of the Action Plan was the requirement to develop biodiversity indicators (surrogate measures for a wider range of biodiversity) to enable timely assessment of conservation progress towards the target. In 2004 a European initiative co-ordinated by the European Environment Agency, SEBI 2010 ('Streamlining European 2010 Biodiversity Indicators'), was launched to develop a first European set of Biodiversity Indicators for 2010 target assessment (European Environment Agency 2007).

Components of biodiversity requiring assessment include trends in the abundance and distribution of species. Unfortunately, at a European scale the development of species indicators is problematic because systematic monitoring of biodiversity is scant, with birds providing the best available dataset. Due to the establishment of butterfly monitoring schemes in a number of European countries in recent years that collect annual data to a scientific standard over a wide geographical area, population trends of butterflies now represent an important new possibility as an indicator.

Butterflies are considered as important components of biodiversity because they have considerable resonance with both the general public and decision-makers (Kühn et al., 2008).

Information of trends in butterflies is increasingly used by a number of Northwest European governments. For example, in 2005 the English Government used three butterfly indicators, including a Headline Indicator Populations of Butterflies, and Populations of both Woodland and Farmland Butterflies, to help assess progress in implementing the England Biodiversity Strategy and assessing the effectiveness of biodiversity conservation policies (Department for Environment, Food and Rural Affairs 2006). In the Netherlands, butterflies are included in a headline indicator based on the Red List status of species as well included in several other indicators, e.g. to show effects of climate change (www.natuurcompendium.nl).

Butterflies have been proposed as biodiversity indicators due to their rapid and sensitive responses to subtle habitat and climatic changes and as representatives for the diversity and responses of other wildlife, especially insects (Rosenberg et al. 1986, Erhardt & Thomas, 1991, Fleishman et al., 2000, Kremen 1992, New et al. 1995, Hammond, 1995, Beccaloni and Gaston 1995, Oostermeijer and van Swaay 1998, Ehrlich 2001, Ehrlich 2003, Parmesan, 2003, Thomas 2005). Representation for insects would be particularly important as insects comprise 56% of known species (Groombridge 1992) and an estimated 80% of the global species stock (Stork 1993).

In this paper, we evaluate the suitability of butterfly population data as a biodiversity indicator at a European scale for 2010 target assessment. We discuss the strengths and weaknesses of the preliminary grassland European Butterfly Indicator as reported by Van Swaay & Van Strien (2005), and compare this indicator with the farmland bird indicator as developed by Gregory et al. (2005). We also discuss briefly how well trends in butterflies may represent trends in other insects groups. Grasslands are vitally important to European butterflies, providing habitat for 88% of species and the main habitat for 88% of species (Blab and Kudrna, 1982, Tax, 1990, Van Swaay & Warren, 1999; Van Swaay et al., 2006). In many cases grassland butterflies are dependent on agricultural management for their long-term survival. Thus there are strong linkages to EU policy mechanisms such as the Common Agricultural Policy and agri-environment schemes.

Methods

Evaluation of using butterfly monitoring data as an EU Biodiversity Indicator

The potential use of butterfly monitoring data in a European indicator was evaluated in two ways. First, by applying the following criteria to butterfly monitoring data: policy relevance, biodiversity relevance, scientifically sound and well founded methodologically, broad acceptance and understandability, affordable monitoring, available and routinely collected data, affordable modelling, spatial and temporal coverage of data, representativeness of the data and sensitivity. These criteria were developed and applied by the European Environment Agency (EEA, 2007). The quality results for each criterion were scored on a scale from 0 (no score) to 3 (highest score), with the total enabling objective comparison with other candidate indicators.
Secondly, a trial indicator for grassland butterflies was made. This provides practical insights into the strengths and weaknesses of the monitoring data and methods.

*Table 7.1: Characteristics of the Butterfly Monitoring Schemes. The data from countries or regions marked by *were used for the preliminary European Butterfly Indicator. [1)] after weighting, see chapter 6. ** Assessed by expert*

	Starting year (-End Year)	Number of transects per year	Number of visits on a transect per year	Field work by (v=volunteers, p=professionals)	Method to choose transects	Representative for grasslands in the wider countryside**	Nature reserves over-represented in the indices**
Belgium – Flanders *	1991	10-20	15-20	v	free	no	no
Estonia	2004	7-10	9	p	by co-ordinator	no	no
Finland *	1999	50-60	10-16	v	free	yes	no
France	2005	75	4-8	v	random	yes	no
France – Doubs *	2001 (-2004)	10	10-15	p	by co-ordinator	yes	no
Germany	2005	400	15-20	v	free	yes	yes
Germany - Nordrhein Westfalen *	2001	50	15-20	v	free	no	yes
Germany – Pfalz * (*Maculinea nausithous* only)	1989 (-2002)	16	3	p	by co-ordinator	yes	no
Jersey	2004	15	15-25	v	free	yes	no
Spain – Catalunya *	1994	60-70	30	v	free	yes	no
Switzerland – Aargau *	2001	100	4-7	p	systematic	yes	no
The Netherlands *	1990	430	15-20	v	free	yes	no [1)]
Ukraine – Transcarpathia *	1974	60	2	p	free	yes	no
United Kingdom *	1976	750	15-20	v	free	?	yes

Collation of butterfly monitoring data from European schemes

Regional and national butterfly trend data were collated through Dutch Butterfly Conservation/Butterfly Conservation Europe from a consortium of individuals and organizations from nine countries including: UK, Ukraine, Germany, Netherlands, Flanders (Belgium), Spain, Switzerland, Finland, and France (Table 7.1).

Butterfly monitoring methods

The main objective of European Butterfly Monitoring schemes is to assess changes in abundance at national and regional levels of butterflies, including Habitat Directive species. For the bulk of schemes the field method used closely follows that developed for the British Butterfly Monitoring Scheme, established in 1976 (Pollard & Yates 1993). Counts are made in a fixed area along line transects under set weather conditions and time of day criteria. Counts are made on a regular basis over the flight season of the species monitored and used to generate annual indices for each species at each site. The average number of visits per year varied considerably across the schemes (Table 7.1). Most of the transects are recorded by skilled volunteers, who have a good knowledge of the transect butterfly fauna and their results are checked by butterfly experts. In many national schemes, transect locations are not randomly selected, but are based on free choice of volunteers (Table 7.1). This may easily lead to oversampling of semi-natural grasslands, nature reserves and other protected areas and under-sampling of intensive grasslands on privately owned farmland in the wider countryside.

Preliminary European Butterfly Indicator: habitat and species selection

The habitat focus was grassland, as this is probably the single most important broad habitat type for butterflies in Europe (Van Swaay et al., 2006). Using widely accepted definitions (e.g. Asher et al. 2001) derived from autecological studies, grassland butterflies were grouped into two broad types: widespread species (mobile species occurring in a diverse range of grassland types) and specialists (low mobility species restricted to semi-natural grasslands). A selection of 17 species was made by European butterfly experts of species that were considered to be characteristic of European grassland using the following criteria: (1) widespread across Europe, (2) sampled by the majority of Butterfly Monitoring Schemes and (3) grassland must be their main habitat as defined in Van Swaay et al. (2006). The seven widespread species were *Ochlodes faunus, Anthocharis cardamines, Lycaena phlaeas, Polyommatus icarus, Lasiommata megera, Coenonympha pamphilus and Maniola jurtina. The ten specialist species were Erynnis tages, Thymelicus acteon, Spialia sertorius, Cupido minimus, Maculinea arion, Maculinea nausithous, Polyommatus bellargus, Polyommatus semiargus, Polyommatus coridon* and *Euphydryas aurinia*.

Preliminary European Butterfly Indicator: indices and trends

Development of a preliminary European Butterfly Indicator for grasslands followed methods recently developed for European Birds (Gregory et al. 2005), with the work carried out in close consultation with experts from Statistics Netherlands, the European Topic Centre for Biodiversity and the European Bird Census Council/Birdlife. National indices were produced for each grassland species in each country, using the program TRIM, which models data across sites and years, accounting for missing indices and zero counts by log-linear modelling (Pannekoek & Van Strien 2003). European species trends were then calculated for each species by combining national results, with a weighting procedure accounting for the difference in national population size of each species in each country. As no precise national population estimates were available, the weighting was defined more precisely as the range proportion that each country (or region) held of the

European distribution for each species (Van Strien et al. 2001, Van Swaay & Warren 1999). A further complication as compared to birds is that the count data per site concern several visits per year. The average number of visits of schemes were taken into account in the weighting too. Missing year totals were estimated by TRIM in a way equivalent to imputing missing counts for particular sites within countries (Van Strien et al. 2001). Multi-species indices for all-species, widespread species and specialist grassland butterflies were derived by calculating the geometric mean index across each species assemblage (Gregory et al. 2005). In this, for each year separately, the log of each species index value was taken, then averaged across selected species and the exponential of the result calculated.

Results

Trends in the European Grassland Butterfly Indicator

There was a steep populations decline of about 40% in the European Butterfly Indicator for grassland butterflies since 1990 (Figure 7.1a). Within this trend, the declines of specialist and widespread grassland species did not differ significantly (specialists average trend value -1.56±4.40 year^{-1}; widespread species average trend -1.94±0.47 year^{-1}; t-test p=0,93).
Changes in the grassland Butterfly Indicator were compared to changes in the indicator for European Farmland Birds using data from the Pan-European Common Bird Monitoring Scheme (PECBMS) (Gregory et al. 2007). From 1990 to 2004, farmland birds declined at a rate of about 20%. Although this suggests a steeper decline for butterflies (Figure 7.1b), the average trend value of farmland birds (-0.59±0.59 year^{-1}, n=33 species) did not differ significantly from grassland butterflies (-1.72±2.54 year^{-1}, n=17 species) (t-test p=0.67).

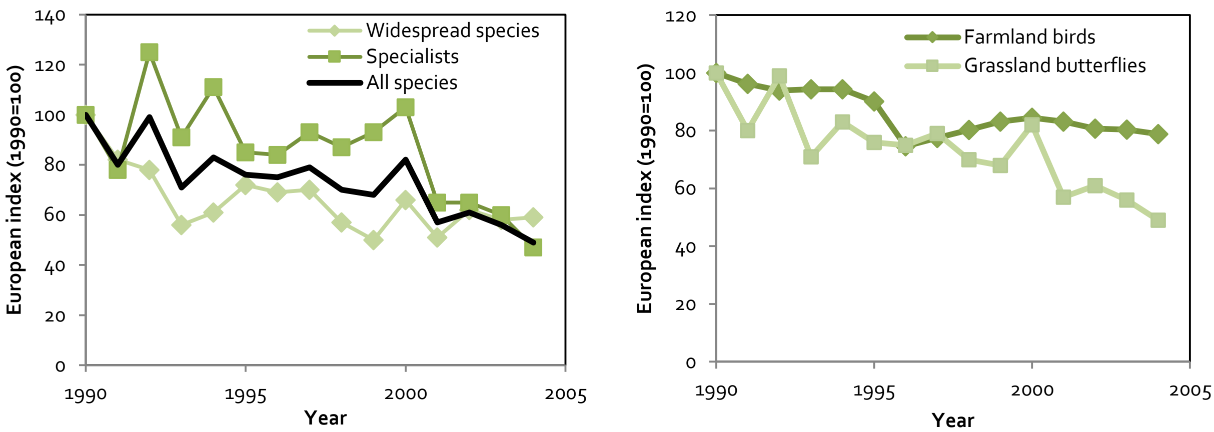

Figure 7.1a (left): Trends in the preliminary grassland butterfly indicator 1990-2004. Figure 8.1b (right): Comparisons of grassland butterflies and farmland bird trends. Bird data source: PECBMS.

Evaluation of using butterfly monitoring data as an EU Biodiversity Indicator

Based on the EEA criteria, overall the butterfly indicator scored highly, validating and confirming the potential of this indicator at a European scale. Policy relevance and biodiversity relevance of the butterfly indicator had the highest score (3); most other aspects had score 2. Spatial and temporal coverage and representativeness had score 1-3, indicating that further improvements are recommended. These last points were confirmed in the trial of the indicator. The spatial coverage is limited to nine countries, mainly in Western Europe (figure 7.1). The temporal coverage is also limited, with the longest time series in the UK, the Netherlands, Catalunya and Transcarpathia (table 7.1). These time series may well be influential for the indicator results. The representativeness of national trends varies across countries, depending on how transects are selected and if any statistical adjustments are made (table 7.1).

Discussion

Trends in the European Grassland Butterfly Indicator

The declining trend in grassland butterflies underlines the policy relevance of a European Butterfly Indicator. Expert opinion predicted this decline, though the rate was more severe than expected. The decrease in grassland butterflies parallels recent historical declines shown though many studies at national (e.g. Asher et al. 2001, WallisDeVries et al. 2002) and international scales (Van Swaay & Warren 1999, Van Swaay et al. 2006). These declines have largely been attributable to habitat loss and modification through agricultural intensification (Asher et al. 2001, Van Swaay et al. 2006), a result largely consistent with studies of other wildlife taxa (Flowerdew 1997, Donald et al. 2001, Robinson & Sutherland 2002, Gregory et al. 2005). In Eastern and Southern Europe abandonment is a serious threat, especially in areas that are too wet, steep, rocky or otherwise unsuitable for intensive farming. Following abandonment, some butterfly species flourish for a few years because of the lack of management, but thereafter scrubs and trees invade and the grassland disappears, including its rich flora and butterfly fauna.
However, inappropriate conservation management (Davies et al. 2007, Konvicka et al. 2005), habitat fragmentation (Thomas 1995, Hanski 2003), and environmental change including climate change (Thomas et al. 2004, Franco et al. 2006) and increased nitrogen deposition (WallisdeVries et al. 2006) may also be important factors in recent declines.

Recent analyses of distribution data from the UK have shown that butterflies are declining in range more rapidly than either birds or plants in Britain (Thomas et al. 2004), emphasising the propensity for butterflies to react more quickly to environmental change than species at higher trophic levels. In contrast, Thomas (2005) has shown that rates of butterfly declines are more comparable to other terrestrial insect groups, although there are examples where this is not the case. Butterflies may respond more rapidly than birds and plants due to their (1) narrow niches, (2) low mobility and (3) their dependence on spatially and temporally dynamically distributed habitats (Thomas et al. 2004).

Comparing changes in the grassland butterfly indicator with changes in the farmland bird indicator suggests that butterflies are declining more rapidly than birds at a supranational level (Figure 7.1b). However, the average trend values between birds and butterflies did not differ significantly. This might be due to the still limited statistical power of butterfly trends (see next section). Also, the bird monitoring data cover a large part of Europe, whereas butterfly data mainly come from the western part of Europe where trends may be more severe than in Eastern Europe. A further point is that the two indicators are not directly comparable. The butterfly indicator chiefly samples butterfly trends on semi-natural grasslands, which are predominant in parts of Central and Eastern Europe but a minority grassland habitat over much of Northwest Europe, whilst the bird indicator is more representative of the whole agricultural landscape, including arable land. Future more sensitive comparisons are required to assess whether birds and butterflies have indeed different trends at a supranational level.

Evaluation of using butterfly monitoring data as an EU Biodiversity Indicator

The Grassland Butterfly Indicator demonstrates how butterflies respond quickly to changes in the environment and how butterflies are thus a good 'early warning' indicator of changes in Europe's biodiversity. The Grassland Butterfly Indicator is disaggregated into (habitat) specialist and widespread species. The specialist index is likely to represent a large amount of biodiversity as habitat specialist butterflies are largely restricted to semi-natural habitats (Asher et. al. 2001), which are among the most species-rich insect/ plant habitats in biodiversity terms in farmland landscapes and are also critically important for rare species (Fry & Londsdale 1991, Thomas 2005). Semi-natural habitats may also be important in maintaining insect diversity in the wider agricultural landscape (Samways 2005, Tscharntke et. al. 2005, Öckinger & Smith 2007).

Butterflies are relatively easy to recognize and data on butterflies have been collected for many years and by thousands of voluntary observers. The method for monitoring butterflies is well described, extensively tested and scientifically sound (Pollard 1977, Pollard & Yates 1993). Following the method used for European birds was technically relatively straightforward - though there were more difficulties to overcome in terms of accounting for the different number of visits between schemes.

Apart from these strengths, several weaknesses should be noted that deserve future improvements. The standard errors of trend estimates of butterflies, especially for specialist species, were considerable and larger than for birds (see the standard errors of the average trend per species group mentioned above), leading to a more fluctuating grassland indicator as compared to the bird indicator. This is caused by the small total number of sampling transects, especially for the rare specialist species, the relatively short time series and the considerable year-to-year fluctuations of species. Low power may limit the opportunity to detect any trend. In practice, however, many trends appeared to be so strong that they were still detectable. The same accounts for the indicator.

There are concerns over the extent to which the trends on butterfly monitored sites reflect trends across the whole European grassland landscape, due to sampling bias. In particular, some butterfly schemes over-sample semi-natural grasslands in nature reserves and other protected areas, and under-sample intensive fields and linear grassland habitats in the wider countryside (Table 7.1). This is a particular problem for reporting on abundance trends of widespread grassland species in Northwest Europe, where the majority of the total population is likely to be located in intensively farmed areas of the wider countryside. However, in the UK, studies have shown that abundance trends in widespread species are extremely similar (1) on semi-natural sites compared to the wider countryside and (2) in protected areas compared to non-protected areas (Brereton & Roy 2006); suggesting that this bias may not necessarily strongly influence national trends. In terms of nature reserves, it has been suggested (Buckland et al. 2005) that butterfly and other species trends may be biased due to more favourable trends on reserves compared to non-reserve land, as the primary objective of land management on reserves is biodiversity conservation. In the Netherlands, grassland butterflies have declined at the same rate in semi-natural grassland nature reserves compared to non-reserve farmland areas (Figure 7.2; paired t-test p=0.86). Studies in the UK that have assessed butterfly trends on reserves have shown that butterflies have performed equally poorly on reserves compared to elsewhere (Thomas 1984, Thomas 1991, Warren 1993, Thomas 1995, McLean et al., 1995, Brereton et al. 2002, Brereton et al. 2007). These results suggest that the suggested bias is not necessarily there.

In the Netherlands, sampling bias (over-sampling of particular habitat types) has been corrected by post-stratification and statistical weighting (Van Swaay et al.

2002). However, if the number of monitored sites is low in habitats that comprise a large proportion of the land surface, it can be dubious to attempt such weighting procedures. For common species monitoring, it is advisable to establish a scheme with a more formal survey design (Yoccoz et al. 2001, Buckland et al. 2005, Legg & Nagy 2006). A number of more recent national butterfly schemes (e.g. in Switzerland and France, and planned in the UK - Roy, Rothery and Brereton 2007) have been designed with a greater emphasis on representative transect selection (based on random sampling) and efficiency savings (fewer visits) (table 7.1). Finally, the coverage across Europe is still limited. It is important that more monitoring is started in as many countries as possible to improve the representativeness of the indicator for Europe as a whole. There are already encouraging developments in this respect, with for example new schemes proposed for Portugal, Ireland and Slovenia.

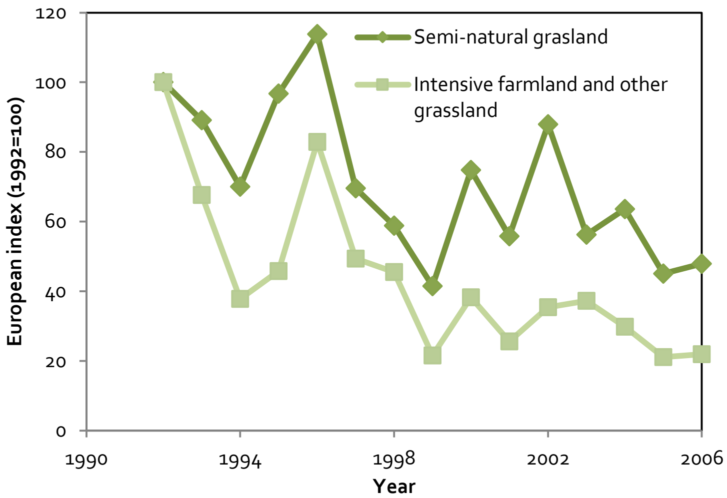

Figure 7.2: Trends in grassland indicators 1992-2006 in semi-natural areas in nature reserves and in farmland areas in the Netherlands. The indicators are based on 15 grassland species. Data are from the Dutch butterfly monitoring scheme. For details see www.natuurcompendium.nl

Butterflies as biodiversity indicators

Butterflies are the only invertebrate taxon for which it is currently possible to estimate rates of decline among terrestrial insects (de Heer et al. 2005, Thomas 2005). However, butterflies can only be regarded as good biodiversity indicators if it is possible to generalise their trends to a broader set of species groups (Pearson 1995, Hilty & Merenlender 2000, Balmford 2002). The distribution of butterflies has been found to be a good predictor of areas of high biodiversity, species richness and or habitat quality in the majority (though not all) of studies (Beccaloni & Gaston 1995, Brown 1991, Brown & Freitas 2000, Simonson et al. 2001, Fleishman et al. 2005, Grill et al. 2005, Kerr et al. 2000, Kremen et al. 2003, Thomas & Clarke, 2004, Maes & van Dyck 2005, Maes et al. 2005, Ricketts et al. 2002).

There is only limited evidence to indicate that changes in butterfly abundance, species-richness and distribution mirror changes in other taxa (Blair, 1999; Swengel & Swengel 1999, Brown & Freitas 2000, Conrad et. al. 2004, Hickling et al., 2006, Thomas & Clarke, 2004, Thomas et al. 2004). However these studies are not fully conclusive and may be dependent on the taxa and the spatial scales considered (Ricketts et al. 2002). A particular problem is a lack of available data on trends in the abundance of other insects for comparison. In the UK, the best available long-term dataset is for moths, through the Rothamsted Insect Survey (Woiwood & Hartington 1994, Conrad et. al. 2004, Conrad et. al. 2006). Although the figures are not directly compatible because of the differing estimation methods, the decline in the composite measure for moth abundance (total catch of n= 337 species) is significantly negatively correlated with the composite measure for butterfly abundance (the UK Butterfly Indicator of n=52 species) (R=0.54, P=0.03, N=27 years, 1976-2002).

Based on a comprehensive review of studies into their life-history traits, biology, relative sensitivity to climate change and adjusted extinction rates, recent reviews (Ehrlich 1994, Ehrlich 2001, Thomas 2005) have concluded that butterflies may be considered reasonable, albeit imperfect representative indicators of trends observed in the majority of other terrestrial insects (excluding for example invertebrate groups that are predominantly predators and parasitoids). We therefore believe they have a valuable role to play in understanding trends in this crucial part of biodiversity and that the greater risk is to exclude an insect indicator altogether. We suggest that adoption of butterflies in the EU Headline suite would complement the European Bird Indicator by providing a more appropriate representation for insects and for species-rich semi-natural habitat fragments.

Next stages

Currently (April 2007) butterflies along with birds have been put forward as one of the 26 indicators in the first 2010 target headline set. In addition to a grassland butterfly indicator, it is proposed to develop also a butterfly indicator for woodlands. This will enable trends in European butterflies to be disaggregated by woodlands and grassland habitats. European butterfly monitoring data may also play a crucial role in assessing: (1) future climate change impacts (2) whether protected areas (e.g. Natura 2000 sites) are being managed appropriately to maintain the full complement of species with differing fine-scale habitat requirements and (3) whether efforts to mitigate against the effects of habitat fragmentation are successful.

8. Differences in the climate debts of birds and butterflies at continental scale

Slightly modified from:
Devictor, V., Van Swaay, C.A.M., Brereton, T., Brotons, L., Chamberlain, D., Heliölä, J., Herrando, S., Julliard, R., Kuussaari, M., Lindström, Å., Reif, J., Roy, D., Schweiger, O., Settele, J., Stefanescu, C., Van Strien, A., Van Turnhout, C., Vermouzek, Z., WallisDeVries, M., Wynhoff, I., Jiguet, F.
Nature Climate Change ***2****: 121–124.*

Abstract

Climate changes have profound effects on the distribution of numerous plant and animal species (Parmesan, 2006; Thomas et al., 2006; Lenoir et al., 2008). However, whether and how different taxonomic groups are able to track climate changes at large spatial scales is still unclear. Here, we measure and compare the climatic debt accumulated by bird and butterfly communities at a European scale over two decades (1990–2008). We quantified the yearly change in community composition in response to climate change for 9,490 bird and 2,130 butterfly communities distributed across Europe (Devictor et al. 2008). We show that changes in community composition are rapid but different between birds and butterflies and equivalent to a 37 and 114 km northward shift in bird and butterfly communities, respectively. We further found that, during the same period, the northward shift in temperature in Europe was even faster, so that the climatic debts of birds and butterflies correspond to a 212 and 135km lag behind climate. Our results indicate both that birds and butterflies do not keep up with temperature increase and the accumulation of different climatic debts for these groups at national and continental scales.

Hipparchia statilinus *is the most warmth-loving species in the Netherlands with the highest Species Temperature Index (STI). It is only found at hot places on bare sand.*

Species are not equally at risk when facing climate change. Several species-specific attributes have been identified as increasing species' vulnerability to climate change, including diets, migratory strategy, main habitat types and ecological specialization (Jiguet et al., 2007; Heikkinen et al., 2010; Warren et al., 2001). Moreover, although phenotypic plasticity may enable some species to respond rapidly and effectively to climate change (Visser, 2008; Charmantier et al., 2008), others may suffer from the induced spatial mismatch and temporal mistiming with their resources (Parmesan, 2007; Sherry et al., 2007). For instance, species such as great tits and flycatchers have been shown to become desynchronized with their main food supply during the nesting season (Visser et al., 1998).
However, beyond individual species' fates, climate change should also affect species interactions and the structure of species assemblages within and across different taxonomic groups over large spatial scales (Schweiger et al., 2008; Harrington et al., 1999; Pounds et al., 2006). For instance, ectotherms should be more directly affected by climate warming and taxonomic groups with short generation time should favour faster evolutionary responses to selective pressures induced by climate changes (Schweiger et al., 2008). Yet, whether different taxonomic groups are tracking climate change at the same rate over large areas is still unclear, and methods to routinely assess the mismatch between temperature increases and biodiversity responses at different spatial scales are still missing (Root et al., 2003).
Here, we used extensive monitoring data of birds and butterflies distributed across Europe to assess whether, regardless of their species-specific characteristics, organisms belonging to a given group are responding more quickly or more slowly than organisms belonging to another group over large areas. We characterized bird and butterfly communities in 9,490 and 2,130 sample sites respectively by their community temperature index (CTI) for each year from 1990 to 2008. The CTI is a simple means to measure the rate of change in community composition in response to temperature change (Devictor et al., 2008). It was recently adopted as an indicator of climate change impact on biodiversity by the pan-European framework supporting the Convention on Biological Diversity (Streamlining European 2010 Biodiversity Indicators).
The CTI reflects the relative composition of high- versus low- temperature dwellers in local communities. High- versus low- temperature dwellers are first differentiated according to their species temperature index (STI). The STI of a given species is simply the average temperature of the species range and is taken as a proxy for species' dependence on temperature. CTI is then calculated, for a given monitored site, as the average of species' STI weighted by species abundances (CTI is thus expressed in degrees Celsius). A temporal increase in CTI directly reflects that the species assemblage of the site is increasingly composed of individuals belonging to species dependent on higher temperature (that is with high STI). This approach enables a comparison of the velocity of changes in communities of a given taxonomic group and of temperature.

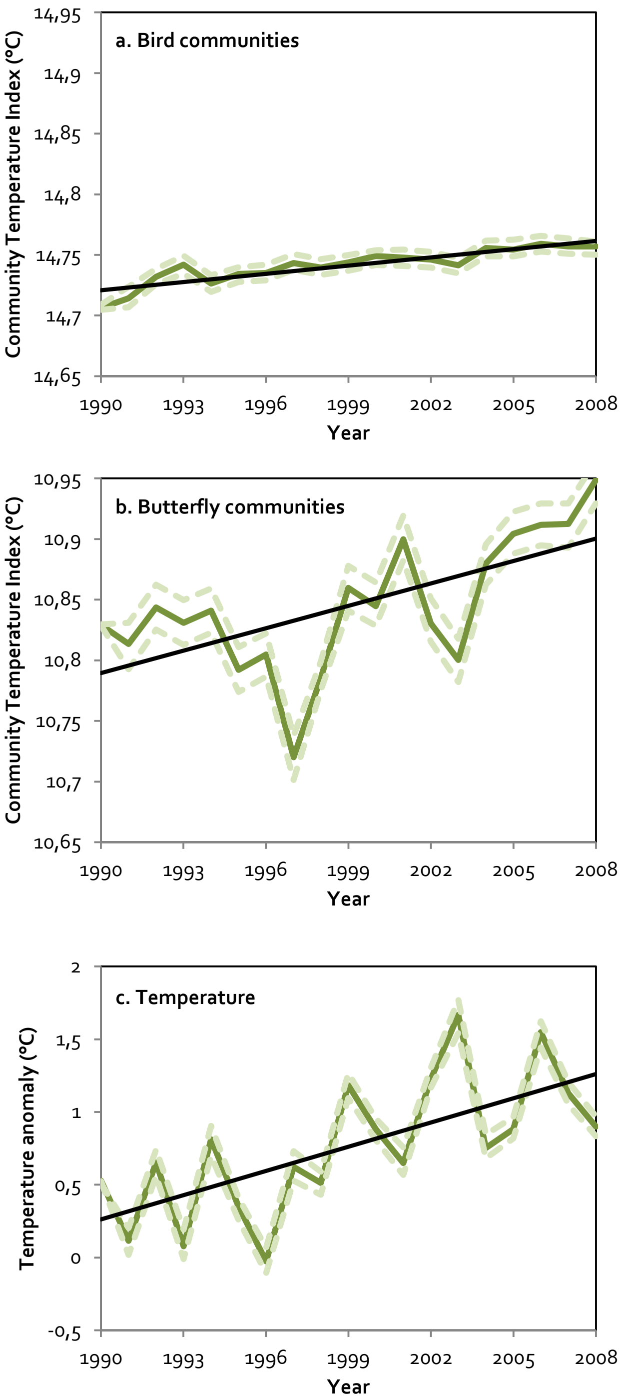

Figure 8.1: Temporal trend of CTI and temperature in Europe from 1990 to 2008 (± standard error of the mean in dashed lines).
a,b: CTI for the bird (a) and butterfly (b) communities monitored in Europe from 1990 to 2008.
c, March–September temperature for the same period. Temperature anomalies are calculated as the departure from the average of the base period 1961–1990.

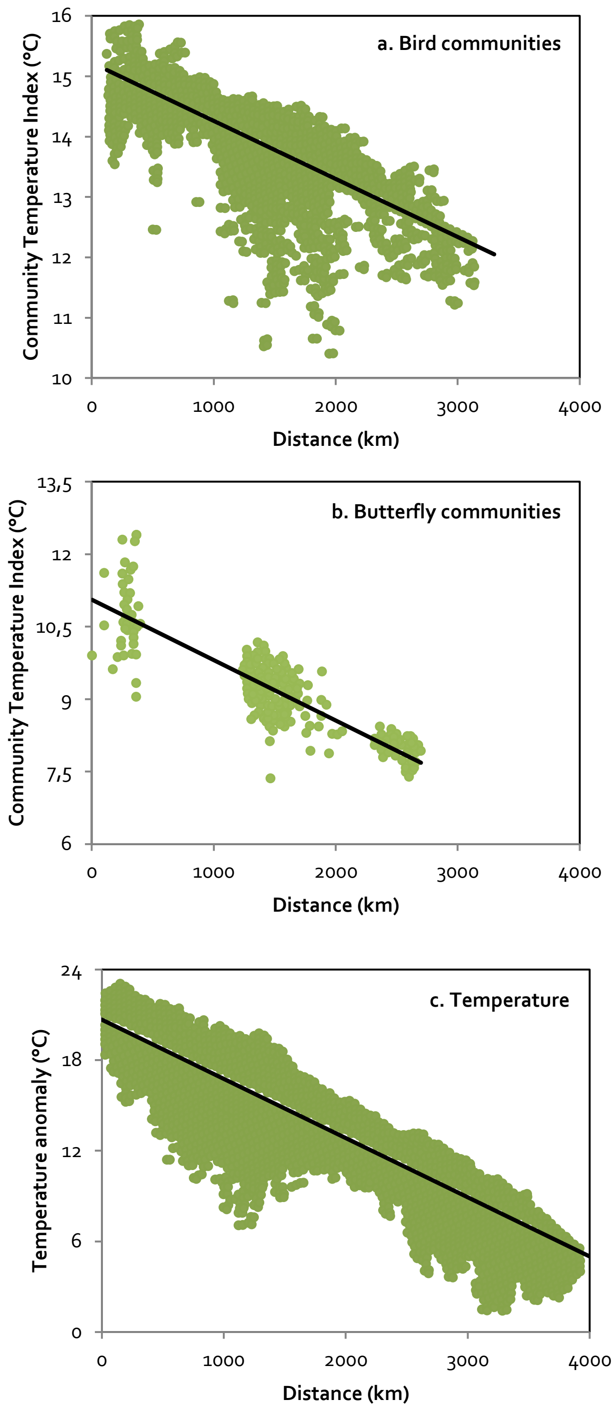

Figure 8.2: Spatial trend of CTI and temperature in Europe.
a,b, Change in CTI for the bird (a) and butterfly (b) communities from south to north.
c, Change in March–September temperature along the same gradient.
For birds and butterflies, each point represents the CTI for a given sample monitored in 2005. Temperature is the average of March–September temperature to match the breeding season of birds and butterflies. Distance (x axis) is calculated from the southern border of the studied region.

Indeed, the temporal slope of the change in CTI gives the rate of change in community composition in response to climate change through time (°C yr^{-1}). The south–north gradient in CTI (°C km^{-1}) then provides an estimate of the rate of change in CTI in kilometres. Providing that this gradient is linear, the temporal change in CTI can be considered as equivalent to a northward shift in CTI using the ratio between the temporal trend and the spatial gradient in CTI (°C yr^{-1}/°C km^{-1} =km yr^{-1}). The same can be done independently for temperature to estimate the velocity of its northward shift (km yr^{-1}; Loarie et al., 2009). The comparison between the velocity of CTI and the velocity of temperature then provides an estimate of the lag between the spatial shift in temperature and community response.

Using this approach, we found that from 1990 to 2008 the CTI of European birds (bird CTI) has increased steadily (+2.6±0.19×10^{-3} yr^{-1}; $F_{1,17}$=92,12; r^2=0.84; P<0.0001; figure 8.1a). Moreover, the CTI spatial gradient is equivalent to a loss of 1.26±0.01x10^{-3} °C of bird CTI each kilometre from south to north ($F_{1,5099}$=4,776; r^2=0.78, P<0.0001; Figure 8.2a). The temporal increase in bird CTI is thus equivalent to a 37±3 km northward shift in the composition of bird communities over the period considered ((2.6×10^{-3}/1.26×10^{-3}) ×18 years).
Using the same approach, we also found that European butterfly communities are increasingly composed of individuals belonging to high-temperature-dwelling species (trend in butterfly CTI: +9.3±0.5×10^{-3} yr^{-1}; $F_{1,17}$=12.6; r^2=0.42; P<0.0001; Figure 8.1b). The temporal trend in butterfly CTI is much steeper than the trend in bird CTI (difference between slopes 6.74±0.5×10^{-3}; P<0.01, analysis of covariance). The spatial gradient in butterfly CTI of 1.47±0.08×10^{-3} °C loss of CTI km^{-1} ($F_{1,797}$=1.748; r^2=0.89; P<0.0001, Figure 8.2b) reveals that the composition of butterfly communities has shifted 114 ± 9 km northward during 1990–2008 ((9.3×10^{-3}/1.47×10^{-3})×18 yr).
During 1990–2008, the temperature also increased steeply (+5.50±0.61×10^{-2} °C yr^{-1}, $F_{1,17}$=79.6; r^2=0.81; P<0.0001; Figure 8.1c). This temporal trend in temperature can be translated in space using the spatial variation of temperature in Europe (Loarie et al., 2009). This gradient is equivalent to a loss of 3.98±0.01×10^{-3} °C km^{-1} from south to north ($F_{1,30674}$=1.7×10^5; r^2=0.84; P<0.00001, figure 8.2c). The temperature increase during 1990-2008 thus corresponds to a northward shift of 249±27 km.
These results indicate that birds and butterflies do not adjust their abundance according to the northward shift of their suitable climates and have accumulated a climatic debt of 212 km and 135km respectively (differences between spatial shift in temperature and in bird CTI and butterfly CTI respectively).
The change in CTI does not tell which and how particular species are affected by climate change but integrates the actual decline of cold species, increase of warm species and the combination of both. Therefore, changes in CTI could mostly result from variations in the dominance structure of species occurring locally rather than from real spatial shifts. However, using presence–absence data rather than abundance, we found similar qualitative results. Therefore, the increase in bird and butterfly CTI also results from changes in the identity of species occurring in local sites rather than only from abundance variations.
Change in CTI could also reflect the strong positive or negative trend of only a few species rather than mirroring profound changes in community composition. To assess whether our conclusions are robust to the identity of the species considered, we used a systematic re-sampling approach in which the trends in the bird and butterfly CTI were estimated after the random removal of 20% of the species monitored in each country. This analysis further confirms the robustness of the findings to the change in the species pool considered.

Climatic debt can be defined as an accumulated delay in species' response to change in temperatures attributable to its inability to track climate change. Our results indicate not only that birds and butterflies are not tracking climate change fast enough at large spatial scale, but also that a lag is expanding between the two groups. Climate change has become a strong selective pressure, and response to this pressure is species and context dependent (Hoffmann et al., 2011). What are the consequences of these increasing climatic debts for each group and between groups at large spatial scale remains to be studied. Genetic variability, population size and generation time, but also dispersal or behavioural plasticity, all contribute to shape species' responses to climate change. In this respect, evolutionary responses to changing climate have already been documented and are particularly expected for short-time generation groups such as butterflies (Skelly et al., 2007). Therefore, significant evolutionary response can, at least to some extent, contribute to the observed trends in CTI.

Although the data we have do not enable us to disentangle the real lag accumulated by birds and butterflies from possible local adaptation to temperature increase, we believe that the rapid adaptations of particular species, if any, are unlikely to produce our results, which are based on many species with likely high variability in their evolutionary response. However, a close inspection of how changes in CTI vary in space or for particular groups of species (defined according to their localization, dispersal ability, genetic diversity, or any trait of interest suspected to induce differential climatic responses between species and/or groups) could possibly help to disentangle evolutionary from demographic processes in the responses. The delay in the climatic debt of bird and butterfly communities may disrupt multiple interactions between species. For example, many bird species depend on caterpillars and could therefore suffer from possible modifications of this direct interaction (Charmantier et al., 2008; Parmesan, 2007; Sherry et al., 2007; Visser, 1998). It is also likely that other groups of terrestrial insects on which many insectivorous vertebrates rely are experiencing important northward shifts and changes in community composition. Moreover, birds and butterflies are among the most dispersive species so they should be able to track climate change more easily than other taxonomic groups. Therefore, other multigroup interactions are also probably facing delayed responses to climate change at large scale with unknown consequences for biodiversity and ecosystem functioning (Parmesan, 2006; Harrington et al., 1999; Memmott et al., 2007). Finally, the negative consequences of such delays are probably enhanced by interacting and self-reinforcing processes between climate and land-use changes (Warren et al., 2001; Brook et al., 2008).
More rapid responses in butterflies than in birds on average (that is, calculated at the European level) may be due to butterflies having relatively short life cycles and being ectothermic, enabling them to track changes in temperature regimes very closely. These differences may induce higher turnover rates in butterfly communities in response to climate changes (Kuussaari et al., 2009; Thomas et al., 2004), which probably contributes to explain the stronger variation in butterfly CTI (Figure 8.1b). Therefore, although birds, as a group, are more dispersive than butterflies, our results suggest that they may accumulate higher climatic debt in the long run.

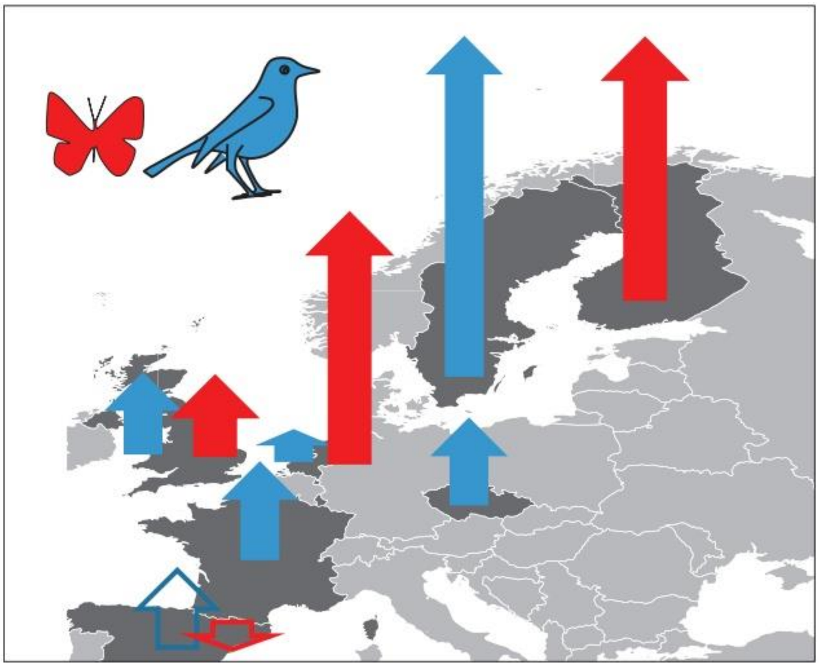

Figure 8.3: European variations in the temporal trend of bird and butterfly CTI. The map shows the temporal trend of bird and butterfly CTI for each country. The height of a given arrow is proportional to the temporal trend per group and its direction corresponds to the sign of the slope (from south to north for positive slopes). The arrow is opaque if the trend is significant.

The ability of each taxonomic group to cope with temperature increase (and hence the potential mismatch between groups) should also depend on the biogeographic, socio-economic and conservation context. When calculated at the country level, we found that the temporal trend in CTI was positive and highly significant within nearly every country. This intra-European analysis also revealed that, for a given taxonomic group, the temporal change in CTI was much faster in some countries than in others (Figure 8.3). For countries with data available simultaneously for birds and butterflies, we found either a much higher trend in CTI for butterflies or no difference among groups. Overall, these results confirm that the compositions of bird and butterfly communities are currently strongly affected by climate change, but also reveal that the differences between groups are dependent on the area considered.

Interestingly, although the magnitude of the CTI is dependent on the number and identity of the species considered, we showed that the detection of a temporal trend in CTI is very robust to changes in the species considered. Indeed, a given change in CTI only reflects the population adjustments of species according to each species-specific thermal distribution, so, in principle, the trend in CTI should remain sensitive to temperature increase whatever the species considered. However, to be meaningful, the CTI must be based on species representing a gradient in STI values. Moreover, the temporal trend in CTI must be calculated on enough sites (and/or years) to avoid confounding factors. Indeed, if the trend in CTI is estimated in a restricted area in which land-use changes have affected a biased sample of species with respect to STIs, the trend could be erroneously interpreted as a community response to climate changes (Clavero et al., 2011).

Understanding the major ongoing changes in structure and composition of communities within and between trophic levels is necessary to prefigure forecasted changes in ecosystem integrity. Future assessments could quantify whether and how potential delays in the response of different taxonomic groups to climate change vary in different habitats and interact with current trends in land-use changes. We therefore suggest that the approach proposed here can help to improve the traceability of climate change impacts on biodiversity in mapping whether, how and where different taxonomic groups are affected by climate changes, using either abundance or presence–absence data, and for national- or international-level assessment.

Methods

We used a method already described to estimate the northward shift in composition of a given taxonomic group (Devictor et al., 2008). In brief, the velocity of bird and butterfly communities and of temperature is obtained in two steps. First, for each taxonomic group, we calculated the annual change in the CTI reflecting the relative composition of high- versus low-temperature dwellers. The CTI is a simple means to measure the rate of change in community composition in response to temperature change. It is calculated, for a given site, as the average of each STI occurring in this site, weighted by the species abundances in this site. The STI of a given species is the long-term average temperature over the species range (CTI is therefore expressed in degrees Celsius). A temporal increase in CTI in a given site directly reflects that the relative abundance of individuals belonging to species dependent on higher temperatures (that is with a high STI) is increasing in this site. We then estimated the overall temporal slope of the change in the pan-European CTI through time separately for birds and butterflies. This trend was estimated using the change in yearly CTI from 1990 to 2008, calculated in 9,490 and 2,130 sample sites (located across Europe from Spain to Finland) respectively for birds and butterflies. These schemes were shown to provide high quality data for building pan-European indicators based on trends in population abundance, and the dataset used in this study represents the largest dataset ever collated documenting temporal changes in the composition of butterfly and bird communities. The slope of this trend gives an estimate of the rate of change in community composition in response to climate change through time (°C yr^{-1}) for each group (Devictor et al., 2008).
Second, we estimated the south–north gradient in bird and butterfly CTI (°C km^{-1}). Because the CTI is linearly decreasing along a south–north gradient, the temporal change in CTI can be considered as equivalent to a northward shift in CTI using the ratio between the temporal trend and the spatial gradient in CTI (°C yr^{-1}/°C km^{-1} =km yr^{-1}). The same was done independently for temperature to estimate the velocity of northward shift in temperature (km yr^{-1}).

Acknowledgements

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Part III: Towards Effective Butterfly Conservation

9. The relationship between butterflies and environmental indicator values: A tool for conservation in a changing landscape

Slightly modified from: Oostermeijer, J.G.B. & Van Swaay, C.A.M.* (1998) Biological Conservation, **86 (3)**, 271-280.*

* these authors contributed equally

Abstract

We examined relationships between Dutch butterfly species and the Ellenberg indicator values for nutrients, acidity and moisture. Presence/absence data on butterflies were obtained from monitoring transects of the Dutch Butterfly Monitoring Scheme. Mean indicator values were calculated from vegetation samples of a selection of transect sections. Single and multiple logistic regression models were used to analyse the relationships. Except for the moisture value, the vegetation samples covered the Ellenberg scales quite well. Significant correlations between moisture and acidity (-) and nutrients and acidity (+) were observed. Sites that were both acid and nutrient-rich were not observed. Most of the observed significant relationships were unimodal (Gaussian), in which species show a clear optimum indicator value. Other species showed a sigmoidal (linear) response to one or more of the ecological indicator values. Several species were significantly correlated with all three indicator values. For a small group this was also the case in the multiple regression model. This was probably caused by multicollinearity of the indicator values, leading to some spurious significant single regression models. We dismiss methodical problems and possibilities for refinements of the models. The observed models can be used to (a) predict the effects of environmental factors on the butterfly fauna, (b) use changes in the abundances of certain species as indicators of ecological processes and (c) determine the sensitivity of butterflies for eutrophication, acidification and ground-water draining. In conclusion, the models provide a powerful aid in the conservation of butterflies in a changing environment.

Pieris rapae: *A butterfly preferring Nitrogen-rich habitats.*

Introduction

The Dutch landscape has been subject to many changes as a result of increasing human disturbance. Among other things, the immense intensification of agriculture of the past decades has caused eutrophication and lowering of the natural ground-water tables of large areas. At the same time, industry and traffic produce compounds that eutrophicate and acidify the environment. As a consequence, many habitats of butterflies have been completely destroyed, and the quality of the remaining habitats is decreasing (e.g. Pavlicek-van Beek et al., 1992; Pullin, 1995). In nature reserves particularly management often has to be intensified to counteract this environmental deterioration and conserve the characteristic species composition (New et al., 1995).
There is a great need to understand and quantify the effects that eutrophication, acidification and lowering of the ground-water table have on wild plant and animal species. When the relationships between various species and environmental parameters can be expressed in the form of models, the effects of environmental scenarios on flora and fauna can be predicted (Latour et al., 1994). In The Netherlands, successful efforts have been made to quantify and model the relationships between plant species and the abiotic environment (Gremmen et al., 1990; Latour and Reiling, 1993; Latour et al., 1994). Using reciprocal averaging, this project has resulted in a calibration of the well-known ecological indicator values of Ellenberg (1979) for most members of the Dutch flora. Moreover, for many species, significant response curves for Ellenberg's nutrient acidity, and moisture values were obtained. The results were used to develop the so-called *M*ultistress m*O*del for the *VE*getation (MOVE: Latour and Reiling, 1993; Latour et al., 1994).
The aim of the study presented here was to quantify the relationships between the butterfly species that occur in The Netherlands and Ellenberg's environmental indicator values for nutrient richness, acidity (pH) and soil moisture, and to consider the usefulness of this approach for its original purpose and for the management and conservation of butterflies.

Methods

Butterfly data

The data from the yearly transect counts of the Dutch Butterfly Monitoring Scheme provided reliable presence/absence data of species on specific locations. The structure of this monitoring scheme is similar to that described by Pollard and Yates (1993). We made a selection of monitoring transects to create a dataset in which different habitat types and regions were represented as equally as possible. It was not possible to achieve a completely balanced set of data, in which, for example nutrient-poor peat bogs were represented equally as well as nutrient-rich agricultural or urban sites. At present, peat bogs are quite rare in The Netherlands and hence will always be under-represented.
The weekly counts of butterflies at each of the selected transects were transformed to presence/absence data. Each transect comprises 8-20 sections of 50m. Sections are generally homogeneous concerning ecotope and management. An overview of the number of sections that represent different ecotopes in the dataset is given in Table 9.1. Data from three consecutive years (1992, 1993 and 1994) were used separately in the analyses to reduce the chance of missing rare or migratory species.

Table 9.1: Distribution of vegetation relevés collected for this study (N = 954) over the various Dutch ecotopes.

Habitat type	Number of vegetation relevés
Woodlands, brushwood, etc.	217
Marsh forest	3
Coniferous and mixed forests	16
Dry deciduous forests	48
Moist deciduous forests	42
Brushwood	46
Coppicewood	4
Wooded banks, dykes, embankments, etc.	45
Scrub	5
Clearcuttings, windthrows and burnt forests	8
Open areas without agricultural use	384
Dune areas	52
Tidal areas	3
Heathlands	76
Semi-natural grasslands	228
Calcareous grasslands	5
Wet hay meadows	7
Semi-natural, moist to wet, poor grasslands	115
Grass heaths	38
Semi-natural dry poor grasslands	50
Raised peat bog areas	19
Rich fen and mire areas	5
Non-linear open waters	4
Large, artificial lakes	1
Small marshes	3
Agricultural areas	41
Grasslands	28
Arable fields	13
Urban areas	38
Ruderal areas	14
Built-on areas	24
Infrastructure	238
Road verges and parking areas	179
Railways and harbour or dock systems	22
Dykes	27
Slopes of canals, waterworks, etc.	10
Linear open waters	22
Ditches, sides of ditches and trenches	22
Shores and banks	10
Shores of peat-bog and turbary lakes	3
Shores of small artificial lakes	1
Banks of rivers, brooks, etc.	6

Table 9.2: The meaning of Ellenberg's indicator numbers for Central European plan species' responses to variation in soil nutrients, acidity and moisture (adapted from Ellenberg, 1979).

	Nutrient number (Stickstoffzahl)	Acidity number (Reaktionszahl)	Moisture number (Feuchtezahl)
1	Very poor	Very acid	Extremely dry
3	Poor	Acid	Dry
5	Moderately rich or poor	Weakly acid	Intermediate
7	Rich	Neutral	Moist
8	Very rich (nitrogen indicator)		
9	Extremely rich (indicating pollution, manure deposits, etc.)	Basic	Wet
10			Frequently inundated
11			Amphibic
12			Aquatic

Environmental variables

In 1996, Braun-Blanquet-type vegetation samples (relevés) were taken at representative sites in individual 50m sections of the selected transects. The vegetation was analysed in the course of one summer in a total of 954 sections from 228 monitoring transects.

The nutrient, acidity and moisture levels of the soil at each butterfly sampling site were inferred from the vegetation composition and the Ellenberg nutrient acidity, and moisture values of individual plant species (Ellenberg, 1979; Melman et al., 1988; Ellenberg et al., 1991). The three Ellenberg scales are explained in Table 9.2. Using the list of plant species compiled for the vegetation relevé, the mean of each of the three Ellenberg indicator values was calculated. Species that are indifferent to a given environmental parameter (category X in Ellenberg's system), or for which the relationship is unknown (?), were excluded from the calculation.

Statistical analysis

The relationships between the presence of butterfly species and the three environmental parameters were investigated using logistic regression analyses (Ter Braak and Looman, 1986; Jongman et al., 1987). The basic hypothesis of the statistical analyses was that the butterfly-environment relationships would have the shape of a Gaussian or unimodal response curve (Figure 9.1; Ter Braak and Looman, 1986; Jongman et al., 1987).

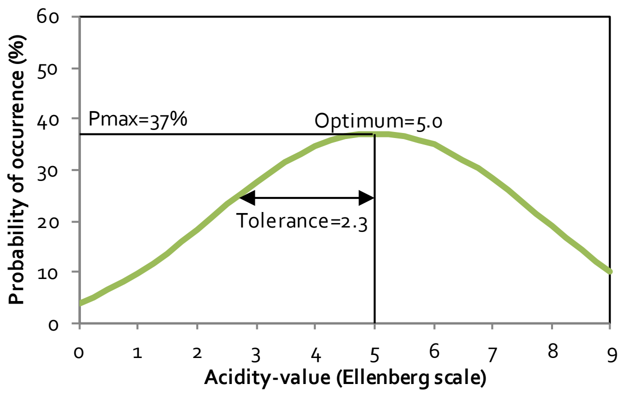

Figure 9.1: Response curve of** Araschnia levana **for Ellenberg's acidity-value, showing the Optimum (U), the maximum probability of occurrence (P_{max}) and the Tolerance (T).

In this model, the probability of observing a butterfly species is related to the Ellenberg value via Eq. (1). In the cases where species occur mainly at one of the extremes of the Ellenberg scale, this Gaussian curve attains the shape of a sigmoidal, often nearly linear, response. If the b_2 term of the unimodal regression model is zero or significantly positive, this suggests a linear relationship (we considered a bimodal response ($b_2 > 0$) ecologically unlikely). In such cases, the sigmoidal model given in Eq. (2) was tested as an alternative hypothesis:

$$p = \frac{e^{b_0+b_1x+b_2x^2}}{1+e^{b_0+b_1x+b_2x^2}} \qquad \text{(Eq. 1)}$$

$$p = \frac{e^{b_0+b_1x}}{1+e^{b_0+b_1x}} \qquad \text{(Eq. 2)}$$

Three different parameters were calculated from the significant Gaussian regression curve (see Figure 9.1) using the method described in Jongman et al. (1987):

1. the 'Optimum' (*U*): the Ellenberg value corresponding with the maximum point of the curve
2. the 'P_{max}': the (maximal) probability of observing a butterfly species at its optimum
3. the 'Tolerance' (*T*): half the width of the bellshaped curve, between the optimum and the point of inflexion, which is a measure of the butterfly's ecological amplitude for the environmental parameter. The range of values at which a species occurs is approximately given by 4T.

Whether the regression parameters, namely the constant (b_0), the linear coefficient (b_1) and the quadratic coefficient (b_2), departed significantly from zero was tested by means of a Wald chi-squared test.

There is a risk that a significant regression between the presence of a given butterfly species and an indicator value is indirectly caused by a (stronger) relationship with another parameter. This risk increases if there are strongly significant correlations between the three ecological indicator values. To check this, we computed Pearson's product-moment correlation coefficients (Sokal and Rohlf, 1988).

To further address multicollinearity, we also performed a multiple logistic regression to study the relative effect of one parameter while keeping the other two parameters constant (Sokal and Rohlf, 1988). When more than one parameter has a significant contribution to the regression, this approach does not result in a regression curve but in a two- or three-dimensional regression plane or surface.

All statistical analyses were performed with the SASI STAT 6.03 package (SAS Institute Inc., 1988).

Results

Distribution of vegetation samples on the Ellenberg scale

The number of relevés per class for each of the three Ellenberg scales is presented in Figure 9.2. Despite efforts to increase the number of observations at the ends of the scales relative to the centre, the intermediate Ellenberg values were apparently much more common on the monitoring transects.

Sites with an average moisture value of <3 (dry to extremely dry) were completely absent from our samples. This was not expected since we sampled several transects in very dry environments (e.g. inland and coastal sand dunes). Probably, some plants indicating dry sites (e.g. *Spergula morrisonii*, *Teesdalia nudicaulis*, and

Saxifraga tridactylites) were not recorded in the transects because of their early flowering time. The fact that moisture numbers > 8.5 were not available is expected, since the values 9 and 10 represent amphibic to aquatic environments, in which butterflies are rarely observed.

Correlations between Ellenberg values

Looking at the number of samples for each combination of Ellenberg values, it is clear that not all combinations are equally represented in the data. The graph for nutrient and acidity value (Figure 9.2a) demonstrates this most clearly. There are few or no transects with a high nutrient value and a low acidity value, or with a low nutrient value and a high acidity value.
As expected on the basis of the distribution of samples in Figures 9.2a and c, significant correlations were observed between the nutrient and the acidity values (positive: r=0.786, p=0.0001) and between the moisture and acidity values (negative: r=-0.121, p=0.0002), but not between the moisture and nutrient values r=-0.036, p=0.2715).

Single (univariate) regression models

For 49 out of 54 butterfly species that were observed on the monitoring transects in 1992-1994, a significant relationship was observed between their occurrence and one or more Ellenberg values (Table 3). For Ellenberg's nutrient value, 26 significant unimodal and 15 significant sigmoidal regressions were found, while eight species did not show a significant relationship. Likewise, for acidity, 28 regressions were significantly unimodal, 16 were sigmoidal and 5 non-significant. The results for moisture value were strikingly different, with only 13 significant unimodal and 29 significant sigmoidal regressions. Seven species did not show a significant moisture response.

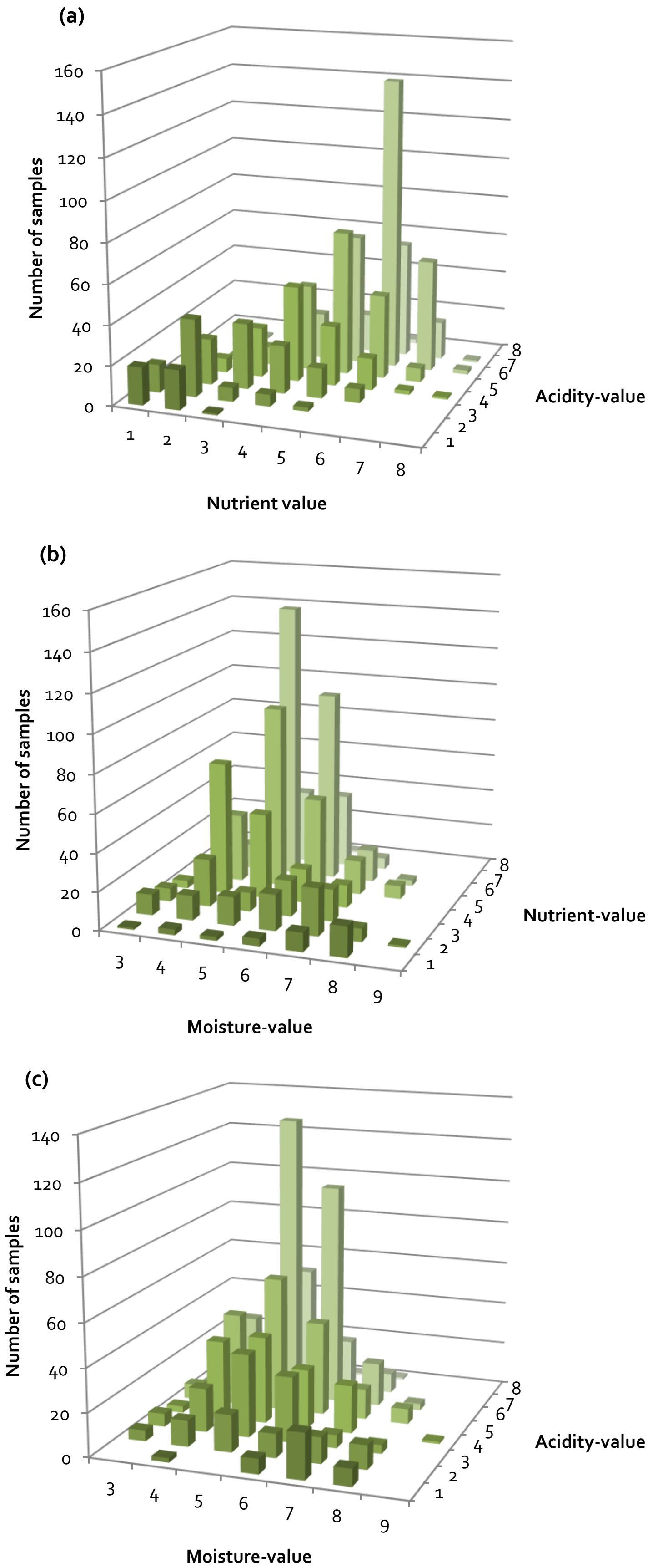

Figure 9.2: Three dimensional diagrams showing the number of vegetation samples for different combinations of Ellenberg values for (a) acidity and nutrient value, (b) moisture and nutrient value and (c) moisture and acidity value.

Table 9.3: Type of univariate response for three of Ellenberg's environmental parameters of 49 butterfly species occurring on Dutch monitoring transects. Uni = significant unimodal (Gaussian) response curve (values of optimum/tolerance between brackets); Sig = significant sigmoidal (linear) response curve (+ or- between brackets indicates a positive or negative slope, respectively); - = no significant response curve.

Species	Nutrients	Acidity	Moisture
Aglais urticae	Sig(+)	Sig(+)	Uni(6.5/2.5)
Anthocharis cardamines	Uni(5.5/2.3)	Uni(5.4/1.8)	Uni(7.3/2.0)
Apatura iris	-	-	Sig(+)
Aphantopus hyperantus	Uni(3.7/3.0)	Uni(3.8/2.2)	Uni(7.0/1.8)
Araschnia levana	Uni(6.4/3.2)	Uni(5.0/2.3)	Uni(6.7/1.7)
Aricia agestis	Uni(5.0/1.8)	Sig(+)	Sig(-)
Boloria aquilonaris	Sig(-)	Sig(-)	Sig(+)
Callophrys rubi	Sig(-)	Sig(-)	Sig(+)
Carterocephalus palaemon	Sig(-)	Uni(2.8/1.5)	Sig(+)
Celastrina argiolus	Sig(+)	-	Sig(+)
Clossiana selene	Uni(3.9/1.2)	Uni(4.9/0.8)	Uni(7.8/0.6)
Coenonympha tullia	Sig(-)	Sig(-)	Sig(+)
Coenonympha pamphilus	Uni(3.7/1.8)	Uni(3.9/2.4)	Sig(-)
Cynthia cardui	Uni(5.6/3.2)	Sig(+)	-
Erynnis tages	-	Sig(+)	Sig(-)
Fabriciana niobe	Uni(3.8/2.0)	-	-
Gonepteryx rhamni	-	Uni(3.0/2.9)	Sig(+)
Heodes tityrus	Uni(2.5/2.1)	Uni(2.1/2.4)	-
Hesperia comma	Sig(-)	Sig(-)	-
Heteropterus morpheus	Uni(3.3/1.0)	Uni(3.2/1.4)	Sig(+)
Hipparchia semele	Uni(2.4/1.9)	Uni(1.8/2.6)	Sig(-)
Inachis io	Uni(5.7/2.8)	Uni(5.9/2.8)	Sig(+)
Issoria lathonia	Uni(4.5/1.3)	Uni(5.4/1.2)	Sig(-)
Ladoga camilla	Uni(4.0/1.2)	Uni(3.6/1.1)	Uni(6.5/0.8)
Lasiommata megera	Uni(5.8/2.3)	Uni(5.9/2.2)	Uni(6.3/1.9)
Lycaena phlaeas	Uni(3.0/2.5)	Uni(2.9/2.6)	Sig(-)
Maculinea alcon	Uni(1.9/0.7)	Sig(-)	Sig(+)
Maculinea teleius	-	-	Sig(+)
Maniola jurtina	Uni(4.2/1.6)	Uni(4.4/2.4)	Sig(-)
Mellicta aurelia	-	Sig(+)	Sig(-)
Mellicta athalia	Uni(2.4/0.6)	Sig(-)	-
Mesoacidalia aglaja	Uni(3.1/0.9)	Uni(2.2/1.4)	Sig(-)
Nordmannia ilicis	Sig(-)	-	Sig(-)
Ochlodes venata	Uni(0.5/3.0)	Uni(1.3/2.7)	Sig(+)
Papilio machaon	-	Sig(+)	Sig(-)
Pararge aegeria	-	Uni(4.0/2.0)	Uni(7.1/1.8)
Pieris napi	Sig(+)	Sig(+)	Uni(6.8/1.6)
Pieris brassicae	Sig(+)	Sig(+)	Uni(6.5/1.9)
Pieris rapae	Sig(+)	Uni(7.5/2.8)	Uni(6.1/1.4)
Plebejus argus	Sig(-)	Sig(-)	Sig(+)
Polygonia c-album	Sig(+)	Uni(5.0/2.2)	Uni(6.5/1.9)
Polyommatus icarus	Uni(4.9/1.6)	Uni(6.2/2.4)	Sig(-)
Pyrgus malvae	Uni(3.2/1.1)	Uni(3.3/1.3)	-
Pyronia tithonus	Uni(3.0/2.7)	Uni(3.3/2.0)	Sig(+)
Quercusia quercus	-	Sig(-)	-
Thymelicus sylvestris	Uni(3.2/2.1)	Uni(3.3/2.1)	Sig(+)
Thymelicus lineola	Uni(5.5/2.3)	Uni(5.5/2.3)	Sig(-)
Vacciniina optilete	Sig(-)	Sig(-)	Sig(+)
Vanessa atalanta	Sig(+)	Uni(6.5/3.5)	Uni(6.4/2.0)

There is much variation in the optima and tolerances among the different butterfly species showing a unimodal response. For nutrient value, the optimum varied between 0.5 and 6.4, while the tolerance ranged from 0.6 to 3.2. The optimum for acidity fell between 1.3 and 7.5. Tolerance for acidity varied between 0.8 and 3.5. In contrast, the range of optimum values for moisture is much smaller: between 6.1 and 7.8.The moisture tolerance varies between 0.6 and 2.5.
A group of rare species characteristically showed relatively narrow tolerances for one or more parameters (tolerance <1.5): *Clossiana selene, Heteropterus morpheus, Issoria lathonia, Ladoga camilla, Maculinea alcon, M. athalia, Mesoacidalia aglaja* and *Pyrgus malvae*. In contrast, the following group of (very) common species demonstrates high tolerances (>2.5): *Aglais urticae, Cynthia cardui, Gonepteryx rhamni, Inachis io, Lycaena phlaeas, Ochlodes venata* and *Vanessa atalanta*.
Figure 9.3 shows the response curves of two species representative of a unimodal and a sigmoidal response. Besides the regression lines, the graphs also show the observed frequencies of observations of the species in the Ellenberg scale classes. One should be aware that logistic regression lines are not simply fitted through these observed frequencies, particularly as the frequencies for low and high values are based on just a few observations, whereas the central ones are based on many observations. Nevertheless, the graphs give some indication of how well the regression lines match the actual observations.

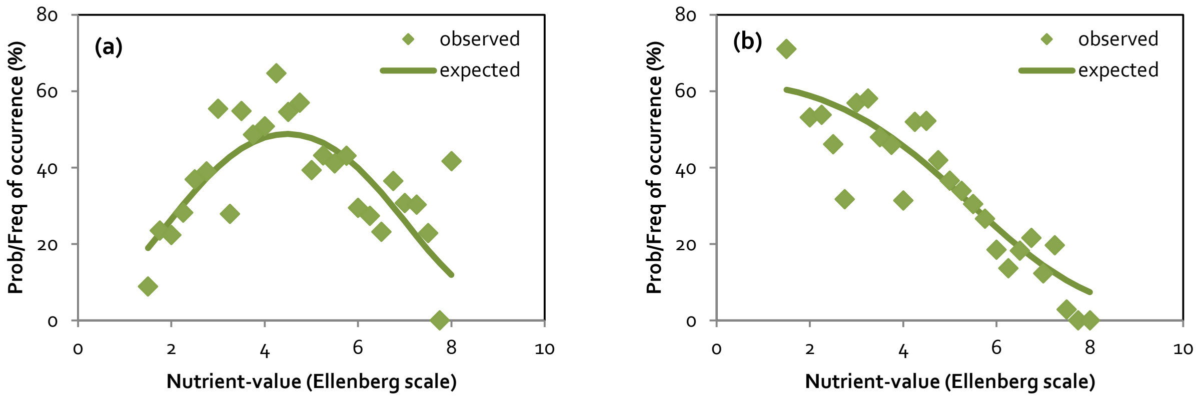

Figure 9.3: Two examples of response curves of butterflies on Ellenberg's nutrient value, showing the calculated logistic regression model (expected) and the observed frequency of the species in the relevés falling in nutrient value classes with a width of 0.25: (a) the unimodal (Gaussian) response of* Thymelicus lineola *and (b) the sigmoidal response of* Ochlodes venata*.

Figure 9.4 gives examples of the wide variety of responses found among different butterfly species. It is important to note that the responses observed agree very well with the responses expected on the basis of the available literature, field experience and 'expert knowledge'. For example, *M. alcon* (Figure 9.4a) has a low P_{max}, for a nutrient value of 12.5% (the probability of observing this rare species is low, even when the nutrient status is optimal), an optimum of 1.9 (it occurs in nutrient-poor wet heathlands and hay meadows), and a very narrow tolerance (it disappears quickly when its habitat is enriched in nutrients, e.g. by atmospheric deposition or the use of fertilisers). On the other hand the more common *Araschnia levana*, whose larvae feed on *Urtica dioica*, has a P_{max} of 35%. This butterfly is mainly found in habitats with a higher nutrient value (optimum of 6.4), but shows a greater tolerance, which agrees with the fact that this species is also observed in more nutrient-poor habitats, as long as there are small patches of *U. dioica* available in the vicinity.

Likewise, the sigmoidal acidity response of *Coenonympha tullia* (Figure 9.4b) with an optimum at very low values (indicating a highly acid environment) and steeply tailing at a value of 3, is not surprising for a species of acid peat bogs. The high P_{max}, and wide acidity tolerance of the very common *Inachis io* is also expected, *Erynnis tages* and *Aricia agestis*, species characteristic in The Netherlands of chalk grasslands and calcareous coastal dunes, respectively, show clear positive sigmoidal responses with a preference for high values (indicating basic soils).

Although only part of the moisture gradient was sampled, *M. alcon* and *Vacciniina optilete* show an expected clear preference for wet habitats (Figure 9.4c), while *E. tages* and *Issoria lathonia* prefer dry sites. The common *Lasiommata megera* has a very high moisture tolerance and is frequent along the entire (sampled) Ellenberg scale.

Multiple regression models

To some degree, multiple logistic regression can alleviate the correlations between the ecological indicator values (Table 9.4). It appears that the species fall into groups, which have significant relationships with similar indicator values. Several species that had significant relationships with all three Ellenberg values appeared to have only one or two significant parameters in the multiple regression model (e.g. *Boloria aquilonaris*, *I. io* and *H. morpheus*). However, the reverse situation (species that were correlated with only one parameter in the single model and with more in the multiple model) was also observed (e.g. *Quercusia quercus*, *Fabriciana niobe* and *Hesperia comma*). Hence, intercorrelation between the environmental parameters clearly had a considerable effect which differs among species.

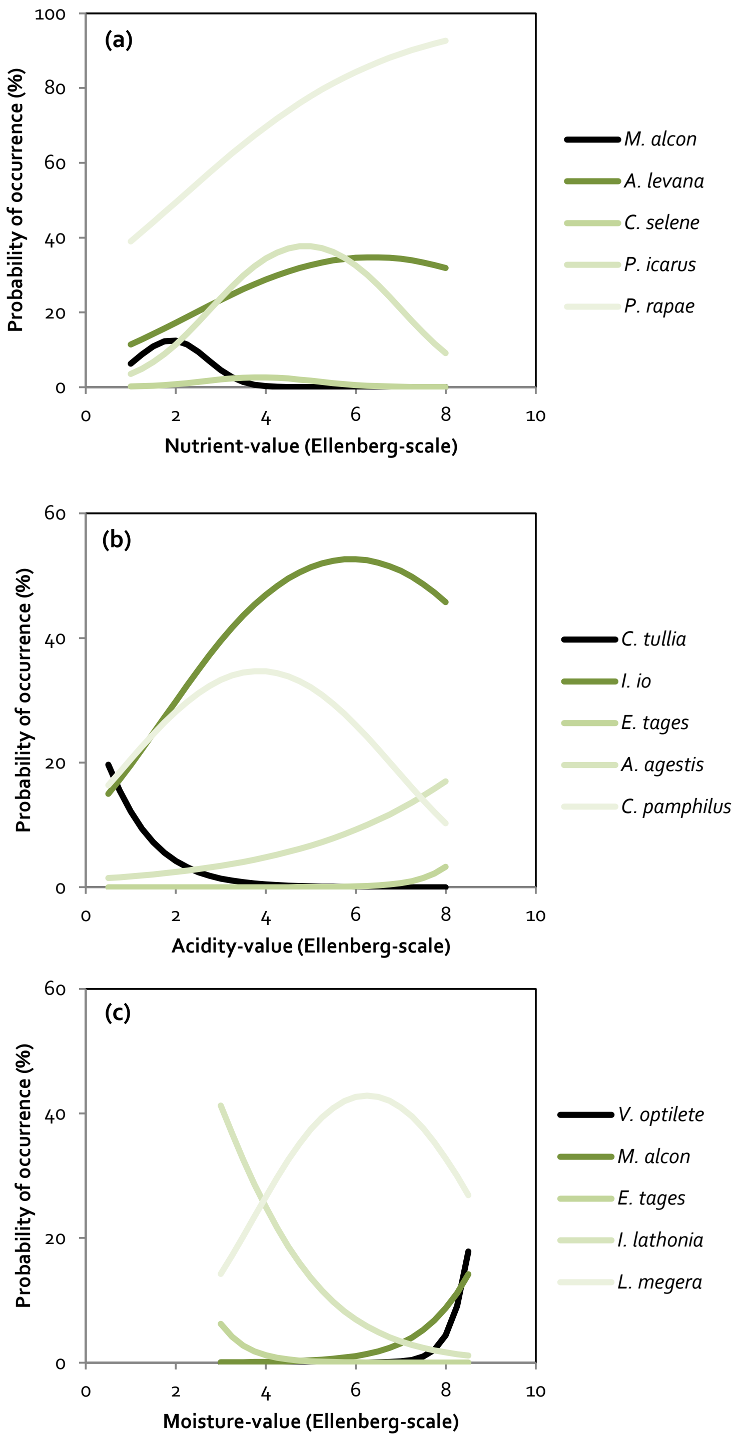

Figure 9.4: Examples of the ecological response curves of various butterfly species for Ellenberg's (a) nutrient, (b) acidity and (c) moisture values. Since the entire scale was not generally sampled, the response curves are only presented for the sampled part.

Table 9.4: Results of multiple logistic regression analysis for butterfly species occurring on the Dutch monitoring transects from 1992 to 1994 with the three Ellenberg indicator values as independant variables; the table shows for which variable(s) the regression coefficients were significantly different from zero (Wald Chi-squared test). N=nutrient value; A=acidity value; M=moisture value. Species for which no variable was significantly different from zero are omitted from this list (11 species). The percentage of explained deviance is calculated according to Oude Voshaar (1994) and Jongman et al. (1987). The species are sorted in groups with the same significant responses.

Species	Significant multiple regression coefficient for	Deviance (%)
Mellicta athalia	N	36
Boloria aquilonaris	N	63
Vanessa atalanta	N,A	2
Maniola jurtina	N,A	1
Thymelicus sylvestris	N,A	5
Polyommatus icarus	N,A	6
Lycaena phlaeas	N,A	4
Thymelicus lineola	N,A	5
Mesoacidalia aglaja	N,M	28
Maculinea alcon	N,M	42
Hipparchia semele	N,M	18
Coenonympha pamphilus	N,M	6
Ladoga camilla	N,M	21
Inachis io	A	2
Polygonia c-album	A	2
Argynnis paphia	A	14
Pieris rapae	A	9
Hesperia comma	A	4
Lasiommata megera	A,M	4
Carterocephalus palaemon	A,M	18
Pararge aegeria	A,M	6
Gonepteryx rhamni	A,M	3
Pieris napi	A,M	8
Aphantopus hyperantus	A,M	5
Araschnia levana	A,M	4
Anthocharis cardamines	A,M	5
Aricia agestis	M	8
Pieris brassicae	M	2
Heteropterus morpheus	M	30
Pyrgus malvae	N,A,M	16
Celastrina argiolus	N,A,M	3
Fabriciana niobe	N,A,M	3
Quercusia quercus	N,A,M	4
Callophrys rubi	N,A,M	38
Ochlodes venata	N,A,M	14
Issoria lathonia	N,A,M	17
Pyronia tithonus	N,A,M	8
Clossiana selene	N,A,M	46

Discussion

General methodical aspects

In principle, any direct relationship between butterflies and Ellenberg values seems unlikely. Because butterflies often have specific host and nectar plants and require certain structural elements for orientation or basking, they would be expected to show a much stronger response to the vegetation at a given site than to the nutrient richness or pH of the soil. However, our models enable a direct quantification of the of 'environmental scenarios' on the butterfly fauna, without the need to make prior predictions about the vegetation.

One important aspect of our models is that we only used butterfly presence/absence data, in spite of the fact that the monitoring data allowed the use of abundances. This meant that an observation of a single individual of a butterfly species on a given transect section was as important as the observation of 50 individuals on another. Obviously, this approach 'throws away' a lot of valuable information in this respect, especially for common and widespread species. On the other hand characteristic species are usually rare, which means the dataset contains a lot of 'zero observations' for which only logistic regression can be used. We therefore decided to use logistic regression for all species so that results could be achieved and compared for as many species as possible.

However, because the available distribution maps of butterflies generally provide only presence/absence data, our models can be used on a wider scale, for instance to predict butterfly distribution on the basis of soil type or for risk assessments, as in Latour et al. (1994). This allows the investigation of the effects of several environmental scenarios (for example the continuation of intensive versus sustainable agriculture) on butterflies by the Dutch government. If we had used abundance data, the models might have been more restricted in their application. Nevertheless models based on abundance data would still be a valuable addition to our knowledge, especially for common and widespread species.

Likewise, the calculation of mean Ellenberg indicator values from the vegetation relevés did not take the abundance of plant species into account. This implies that a few individuals of *U. dioica* (nutrient value=8) with a cover of 1% would contribute equally to the mean nutrient value as a 50% cover of *Succisa pratensis* (nutrient value=2) in the same relevé. In grazed areas, for example, this is a realistic situation, which introduces a possible overestimation of the mean nutrient value of the site. Hence, it could be argued that a weighted calculation of the mean indicator values on the basis of species cover or abundance would have been preferable. On the other hand, from the perspective of *A. levana* or *I. io* the presence of that small patch of *Urtica* in an otherwise nectar-rich pasture may be very important, which suggests that the bias mentioned above was probably realistic within the context of our study. Another argument is that highly indicative plant species are often rare so they could be outweighted by common species with a broad tolerance.

An important factor, which may have a considerable impact on the outcome of the regression analyses, is whether the butterfly and vegetation samples were representative of the Dutch situation. If important habitat types are missing from the dataset, species that occur in these habitats and were only sampled on marginal sites may show an unrealistic ecological response. In this light, the fact that samples with a high nutrient and low acidity value (i.e. acid conditions), or a low nutrient and high acidity value, were not present in the data might have affected the ecological responses of some species. One may ask whether such sites were not sampled or just do not exist. From an examination of the plant species that were assigned an indicator value by Ellenberg, it appears that the second hypothesis is most plausible: plant species with a high nutrient and a low

acidity value are hardly known. There are only two plant species in The Netherlands that meet this criterion (Nutrient value = 8, Acidity value = 3): *Chamerion* (=*Epilobium*) *angustifolium* and *Senecio sylvaticus*. Both species are characteristic of clear-cuttings or open patches in forest types with a rnineralising acid soil. The relationships for some forest butterflies might have been slightly biased because of this.
Since the data were gathered on a national scale, the models do not consider any regional ecotypic differentiation. For example, *P. malvae* occurs in various acid, humid grasslands in the eastern part of The Netherlands, where it uses *Potentilla erecta* as its main host plant. In the coastal dunes, however, it is found in grasslands close to the sea, where *Rubus caesius* is the food plant. In the present study, both habitat types are lumped together in our dataset, which means that the ecological differentiation of the species is averaged. Further analyses will have to deal with this problem because it will undoubtedly affect the accuracy of model predictions. The same holds, of course, for differences on a larger scale. There is increasing evidence, for instance, that butterfly species have different ecological behaviour at the edges of their distribution area (Thomas, 1991, 1993). This means that the relationships of Dutch butterflies presented here cannot simply be translated to other countries.

Single versus multiple regression models
As mentioned in the results, the three environmental parameters for nutrient, acidity and moisture values are not independent. The observed correlations imply that at least some of the significant single regression models may have been caused by a significant relationship with another, intercorrelated, parameter. The multiple regression analyses provided some information about the extent to which these possibly spurious relationships occur in our results. Nevertheless, even though some of the significant single regressions were caused by correlations with one or two other parameters, the information about the optimum, range and tolerance is still useful. This is because the regressions presented here are not describing causal relationships between butterflies and abiotic parameters, but correlative responses with the vegetation as an intermediate, but actually more important, step (invisible in the models). Therefore, in our opinion, both the single and the multiple regressions may be used. The first category is most useful to investigate the sensitivity of species to eutrophication, acidification and lowering of the ground water, because it provides information on the tolerances and ranges, which the multiple regression does not (at least not in a straightforward manner). The second category, however, is more suited for an accurate calculation of the probability of the observation of a species. When the outcome of the multiple regression model is a low probability of observing a species, the single models may help to decide which of the factors is most responsible, especially when more than one environmental parameter contributes significantly to the model.

Practical applications of the results
The models presented here are analogous to the MOVE model for plant species described by Latour and Reiling (1993). The main application of these models is to predict the effects of certain environmental scenarios (changes in environmental processes as a result of political decisions) on flora and fauna. In addition to this, the response curves provide information on the sensitivity of butterfly species to such processes as eutrophication, acidification and lowering of the ground-water table. It is clear that species with very narrow tolerances will be particularly sensitive to changes in the environment. From this study, *C. selene, H. morpheus, I. lathonia, L. camilla, M. alcon, M. athalia, M. aglaja* and *P. malvae* emerge as sensitive species with such narrow tolerances. It is therefore no surprise that these

species are currently all rare and endangered, and have been placed on the Dutch Red List (Wynhoff and Van Swaay, 1995). Knowledge of the ecological responses of these species can be helpful tools for their conservation, as key processes that cause deterioration in their habitat may be identified and countered.

The response curves for butterflies may also be used to evaluate the results of the management of nature reserves directed at specific targets. When nutrient-rich sites are mown annually to develop more species-rich grasslands on poorer soils, the development of the local butterfly fauna may serve as an indicator of the extent of this process. The response curves give information on which species are expected to increase and which to decrease when the nutrient value of the site is lowered by the management. In this respect, some butterfly species are clearly more indicative than others, and can be considered suitable 'process indicators'. Unfortunately, the narrow-tolerance species mentioned above are less suitable indicators because of their rarity. The best process indicators are relatively common species, which show a rather clear response to changes in soil nutrient status, acidity or moisture. Good examples of such species for The Netherlands are *Anthocharis cardamines, Coenonympha pamphilus, Polyommatus icarus* and *Hipparchia semele*.

Another interesting application of the results presented in this paper is the possibility of calculating the suitability of a site for a given butterfly species. Based on vegetation descriptions, the nutrient, acidity and moisture values of a site can be computed. Using the regression models, and especially the multiple models, the probability of occurrence of a butterfly species can be determined and compared with the maximum probability for the species on a national or (in the future) a regional scale. If the probability is still too low, the indicator values of the site will indicate whether a site is too nutrient-rich, too acid or too dry for the species so that the appropriate management actions can be taken. As mentioned above, the single models appear to be very helpful for the latter exercise since these provide information on a species' optimum, tolerance and P_{max}, for a given Ellenberg value.

Of course, several important aspects of butterfly ecology, such as microclimate, vegetation structure or the total area of a given habitat, are not incorporated in our models, which makes predictions of suitability hazardous if no information about these factors is incorporated in the analysis. Nevertheless, the models based on Ellenberg values enable better predictions than when only the ecology of the food plant would have been considered. For example, the acidity value for *Viola palustris*, the major food plant of *C. selene* in The Netherlands, is only 3, while the optimum for the butterfly is 5 (Figure 9.4b). This is probably because the main nectar plants of *C. selene*, such as *Lythrum salicaria* and *Eupatorium cannabinum*, have much higher acidity values. The regression model incorporates transect sections where the adults were observed around the larval food plants, but also sections where they were nectaring. Both environments are very important for the survival of the species, and our regression models integrate them into an estimate of its relationships with the environment.

In conclusion, the methods and models presented in this paper provide a useful tool for the incorporation of butterflies in environmental policies and may help to achieve a more efficient management and conservation of sites for butterfly species in The Netherlands. Several improvements and refinements will be necessary before the models can be used on a wider scale, and we hope that the ideas can also be applied in other countries.

Acknowledgements

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10. Biotope use and trends of European butterflies

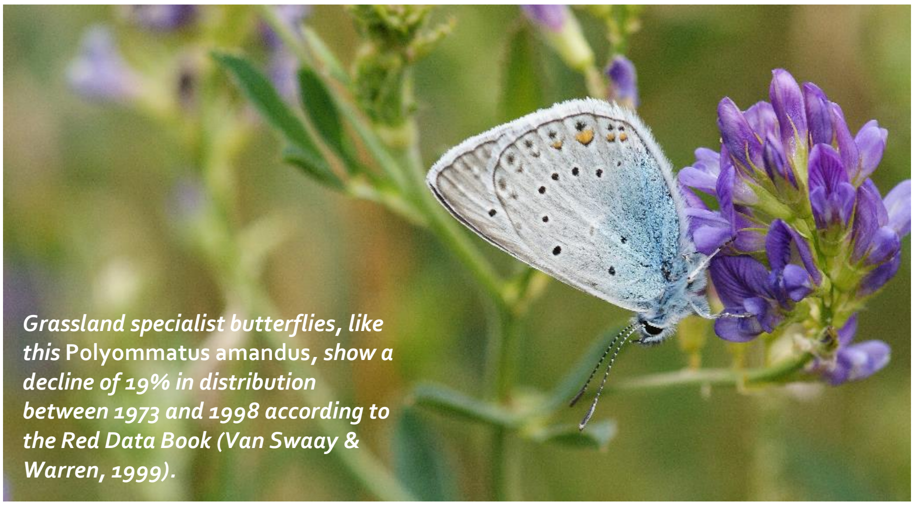
*Slightly modified from: Van Swaay, C., Warren, M., Loïs, G. (2006) Journal of Insect Conservation **10 (2):** 189-209.*

Abstract

Europe has undergone substantial biotope loss and change over the last century and data are needed urgently on the rate of decline in different wildlife groups in order to identify and target conservation measures. However, pan-European data are available for very few taxonomic groups, notably birds. We present here the first overview of trends for an insect group within different biotopes across Europe, based on data from the Red Data Book of European Butterflies.

The most important biotopes for Europe's 576 butterfly species, including threatened species, are man-made or man-influenced, notably types of grassland or heath/scrub communities. Our results show that butterflies are declining substantially across Europe, with a decline in distribution of -11% over the last 25 years. The distributions of the 25 most "generalist" species are declining only slowly (-1%) compared to specialist butterflies of grassland (-19%), wetlands (-15%), and forests (-14%). On average, grassland butterflies have declined somewhat slower than farmland birds (annual decrease –0.8% compared to –1.5%), but woodland butterflies have decreased more rapidly (–0.01% to –0.6%) than woodland birds, which are more or less stable.

The sensitivity of butterflies to environmental changes and the availability of data across Europe suggest that they are very good candidates to build biodiversity indicators and, along with other major groups such as birds, suitable to monitor progress towards the EU target of halting biodiversity loss by 2010.



Grassland specialist butterflies, like this Polyommatus amandus*, show a decline of 19% in distribution between 1973 and 1998 according to the Red Data Book (Van Swaay & Warren, 1999).*

Introduction

Europe has undergone a period of substantial change and development over the last hundred years, which has led to major declines of wildlife and their biotopes in many countries (Horlyck & Lois 2005; Delbaere 1998). However, pan-European data on the rate of decline of species are available for very few taxonomic groups, notably birds (Tucker and Heath 1994; EEA 2004; European Communities 2004; Gregory et al. 2005). Such data are important to properly assess the threats in different biotopes and to identify priorities for conservation action.

Here, we present the first overview of trends for an insect group within different biotopes across Europe, and compare these with bird trends calculated by Birdlife International (Gregory et al. 2005). The analysis is based on data from the first comprehensive review of the status and trends of butterflies across Europe, commissioned by the Council of Europe (Van Swaay and Warren 1999). This showed that butterflies are declining seriously in almost every country and that 71 out of Europe's 576 species are threatened according to the 1994 IUCN criteria (IUCN 1994).

In addition to providing trend data for the Red Data Book, country compilers were asked to provide information on the biotope type used by each species, and the main threats, according to a simple classification system. These results have been used to identify the most important biotopes for European butterflies and to generate trends of species by biotope and identify the importance of biotopes that should be targeted for urgent action. As butterflies have been identified as valuable indicators for many other insects (Thomas 2005), which comprise a large proportion of terrestrial species, we believe the results highlight issues of great importance for the conservation of Europe's biodiversity as well as for assessing European environmental policy. They also demonstrate that butterflies can be used to monitor trends in European biotopes and would provide a valuable and complementary indicator to birds.

Materials and Methods

Red Data Book

Data for the Red Data Book were gathered on all 576 butterfly species known to occur in Europe and were collated primarily by distributing questionnaires to over 50 expert national compilers in all 45 European countries covered by the Council of Europe (Van Swaay and Warren 1999). These data were usually based on the field work carried out by hundreds or even thousands of amateur lepidopterists over many years, often drawing on detailed distribution data.
Using these questionnaires, data were collected on all native species within each country covering:

- Present distribution
- Trend over the last 25 years
- Main biotope used by the species

Species whose ranges just extend within European boundaries, are considered marginal to Europe and were excluded from the review. For all remaining species the European distribution class and trend over the whole continent were calculated, and weighted by country size. Compilers were asked to rank the quality of the trend data from very good, good, moderate, or poor depending on the amount of quantitative data available. These data were used to produce a list of threatened butterflies in Europe, using the 1994 IUCN criteria as closely as possible (IUCN 1994; Van Swaay and Warren 1999).

Each national expert classified the main biotopes for each species in their country according to the main Corine biotope classes, as described in Moss et al. (1991). Their classification was the first attempt to describe European biotopes in a standardized way (Table 10.1).

The nomenclature used follows Karsholt and Razowski (1999), with the exceptions of *Pontia daplidice* and *P. edusa* (summarized as *Pontia daplidice complex*), and *Leptidea sinapis* and *L. reali* (*Leptidea sinapis complex*), since at the time of compilation of the Red Data Book the exact status and distribution and distinction between these species was still unclear.

Biotope profile

A biotope profile was calculated for each species by counting the number of biotope-mentions (= biotope mentioned in a country), and then calculating the percentage of biotope-mentions for each biotope (the biotope profile). Since species with a wide distribution have a long list of biotopes mentioned only once or twice, the biotopes referred to in less than 5% of the biotope-mentions were considered to be of minor importance to the species and were omitted from further analysis. Table 10.2 demonstrates this with the example of *Glaucopsyche alexis*. Biotope data for this Lycaenid butterfly were received from 38 countries. From the 17 listed biotopes, 11 were mentioned only once or twice (less than 5% of the biotope mentions) and were therefore omitted. Consequently the final biotope profile for this butterfly contained only the first six biotope descriptions.

Table 10.1: Classification of the biotopes by Corine biotope descriptions (based on Moss et al., 1991) and grouping to the Main biotope groups.

Corine code	Corine biotope description	Main biotope group
16	coastal sand-dunes and sand beaches	Coastal
18	cliffs and rocky shores	Coastal
31	heath and scrub	Heath and scrub
32	sclerophyllous scrub	Heath and scrub
33	Phrygana	Heath and scrub
34	dry calcareous grasslands and steppes	Grassland
35	dry siliceous grasslands	Grassland
36	alpine and subalpine grasslands	Grassland
37	humid grasslands and tall herb communities	Grassland
38	mesophile grasslands	Grassland
41	broad-leaved deciduous forests	Forest
42	coniferous woodland	Forest
43	mixed woodland	Forest
44	alluvial and very wet forests and brush	Forest
45	broad-leaved evergreen woodland	Forest
51	raised bogs	Wetland
52	blanket bogs	Wetland
53	water-fringe vegetation	Wetland
54	fens, transition mires and springs	Wetland
61	Screes	Unvegetated
62	inland cliffs and exposed rocks	Unvegetated
64	inland sand-dunes	Unvegetated
66	volcanic features	Unvegetated
81	improved grasslands	Agriculture
83	orchards, groves and tree plantations	Agriculture
84	tree lines, hedges, small woods, bocage, parkland dehesa	Agriculture
85	urban parks and large gardens	Urban
86	towns, villages, industrial sites	Urban
87	fallow land, waste places	Urban

Threats

Data on suspected threats were collected only for the 71 European threatened species (Van Swaay and Warren 1999). Fourteen types of threat have been distinguished. National experts have indicated the degree of threat for each threatened butterfly in their country (1=low, 2=medium, 3=high). To calculate the average degree of threat per main biotope type, each threatened species is assigned to the biotope type where it has been mentioned most frequently. This was only possible for forests, grasslands and wetlands. Threats mentioned less than three times have been omitted. Of course, there is a strong risk that biotopes where no endangered species occur are also threatened. Here, the lack of data makes such an assessment unfeasible.

***Table 10.2: Classification of the biotopes of the Lycaenid butterfly* Glaucopsyche alexis.**

Biotope description	Number of mentions	Percentage	Class in table
dry calcareous grasslands and steppes	11	18.3	2
mesophile grasslands	11	18.3	2
broad-leaved deciduous forests	8	13.3	2
dry siliceous grasslands	8	13.3	2
fallow land, waste places	3	5.0	1
sclerophyllous scrub	3	5.0	1
alpine and subalpine grasslands	2	3.3	Not used
heath and scrub	2	3.3	Not used
mixed woodland	2	3.3	Not used
orchards, groves and tree plantations	2	3.3	Not used
Phrygana	2	3.3	Not used
coniferous woodland	1	1.7	Not used
humid grasslands and tall herb communities	1	1.7	Not used
inland rocks, screes and sands	1	1.7	Not used
inland sand-dunes	1	1.7	Not used
tree lines, hedges, small woods, bocage, parkland dehesa	1	1.7	Not used
urban parks and large gardens	1	1.7	Not used

Biotope specialist butterflies

A biotope specialist species was defined as being mentioned more often in one biotope than in the sum of all the others. The following procedure was used to determine the number of biotope specialist species per biotope type per country:

- In order to remove any bias in biotope assessment amongst country compilers, we only included species for which we had biotope data from at least three separate sources, usually from three countries.
- For each species the number of Corine biotopes mentioned per country per species is counted.
- Then, the number of each Corine biotope-mentions per country per species per biotope type is counted.
- These numbers are then evaluated using broad biotope classes (see Table 10.1).
- The percentage of broad biotope classes mentions per biotope type available in the country is calculated for each species.
- Species for which one biotope gets a percentage as high as 50% were considered specialists of that biotope (Appendix 10.2).

Generalist butterflies
To define generalists, each butterfly species was ranked according to the average number of biotopes that it was reported to use compared to the maximum number of biotopes mentioned per country. This allowed the full list of species to be sorted from generalists to specialists. Then, to determine the group of generalists, the top 25 were selected. A control was made on species distribution to avoid narrowly distributed species that would not be representative at the continental scale. The number of countries in which each species occurred was extracted. The method above favours widespread species, and the species selected occurred in a minimum of 18 countries. Nevertheless, this means that especially south European countries were excluded from analysis since in many of these countries availability of good trend data is poor.

The results in Table 10.3 shows that some of the species selected as generalists at a pan-European level are specialists in some parts of their range, especially at the edge of their distribution (e.g., *Pyrgus malvae* and *Papilio machaon*). In this analysis, the definition of "generalist" species thus focuses on the most widespread species that occur in a wide range of biotope types. *Vanessa atalanta* was excluded as it is a migrant species in most of Central and Northern Europe and trends were not available in every country.

Table 10.3: List of butterflies considered to be generalist species at a European level.

Species
Aglais urticae, Maniola jurtina, Anthocharis cardamines, Melanargia galathea, Aphantopus hyperantus, Ochlodes venata, Callophrys rubi, Papilio machaon, Coenonympha pamphilus, Pieris brassicae, Erebia medusa, Pieris napi, Gonepteryx rhamni, Pieris rapae, Inachis io, Polygonia c-album, Iphiclides podalirius, Polyommatus icarus, Issoria lathonia, Pontia daplidice complex, Leptidea sinapis complex, Pyrgus malvae, Lycaena phlaeas, Thymelicus lineola, Thymelicus sylvestris.

Calculating European trends for specialists and generalists
As the quality and accuracy of trend data available from the Red Data Book varied considerably among countries and species, we calculated trends only from those countries that fulfilled the following arbitrary requirements considered to ensure good data quality:

- at least 80% of the species were given a trend, since this shows that sufficient expertise is available and
- not more than 75% of the trends given were "stable" or "fluctuating" as such a high proportion of these categories, often given by default, might be related to a lack of knowledge of national populations especially over such a long time.

This left 20 countries representing more than 50% of Continental Europe area (See Table 10.4, note that Russia and Turkey are excluded here).

Table 10.4: Countries selected for the calculation of trends of specialist and generalist butterflies.

Country
Austria, Belgium, Canary islands, Czech Republic, Denmark, Finland, Germany, Hungary, Latvia, Lithuania, Luxembourg, Moldova, Netherlands, Poland, Romania, Slovakia, Slovenia, Sweden, Switzerland, United Kingdom.

Overall European trends per biotope were obtained as follows:

- Trend classes were converted into trends using the geometric mean of the class extremes. "Extinct" was converted arbitrary to a 99.9% decrease.
- For each species, we estimated the weighted geometric mean and variance, weighted by country area in relation to the mid values of the distribution area occupied within each country (each country compiler classified species along 4 classes of country occupation : <1%, 1-5%, 5-15%, >15%).
- We estimated geometric mean and variance (and thus standard errors) of species according to their attributed biotope group. As a reference group, we also provide the average trend of all the species together to allow a general overview of the situation.

Table 10.5: Total number of species, number of threatened species and the percentage of threatened species per CORINE-biotope. N = total number of species, T = total number of threatened species, %T= percentage threatened.

CORINE-biotope	N	T	T (%)
blanket bogs	45	14	31.1
raised bogs	48	13	27.1
fens, transition mires and springs	59	15	25.4
water-fringe vegetation	75	15	20.0
mesophile grasslands	223	39	17.5
humid grasslands and tall herb communities	171	27	15.8
mixed woodland	187	29	15.5
alluvial and very wet forests and brush	100	15	15.0
coniferous woodland	156	23	14.7
dry calcareous grasslands and steppes	274	37	13.5
broad-leaved deciduous forests	186	25	13.4
heath and scrub	189	25	13.2
alpine and subalpine grasslands	261	34	13.0
dry siliceous grasslands	220	27	12.3
inland sand-dunes	43	5	11.6
broad-leaved evergreen woodland	67	6	9.0
inland cliffs and exposed rocks	70	6	8.6
tree lines, hedges, small woods, bocage, parkland dehesa	128	11	8.6
Phrygana	137	11	8.0
Screes	88	7	8.0
fallow land, waste places	104	8	7.7
orchards, groves and tree plantations	95	6	6.3
cliffs and rocky shores	17	1	5.9
sclerophyllous scrub	202	12	5.9
urban parks and large gardens	96	5	5.2
coastal sand-dunes and sand beaches	40	2	5.0
scrub and grassland	28	1	3.6
towns, villages, industrial sites	66	2	3.0
improved grasslands	74	1	1.4

* three SPEC1-3 species on the Azores (*Hipparchia miguelensis*, *H. occidentalis* and *H. azorina*) are mentioned for agricultural land and artificial landscapes but are not given in the table.

Results

Biotope use

The main biotopes for 436 European butterfly species, based on data collected for the Red Data Book of European Butterflies, are shown in Appendix 10.1 and a summary of the importance of each biotope is shown in Table 10.5.

The results show that the most species-rich biotopes in Europe are dry grassland: notably dry calcareous grasslands and steppes (274 species), alpine and subalpine grasslands (261), mesophile grasslands (223), dry siliceous grasslands (220 species). Mesophile grasslands are also species-rich with 223 species, followed by sclerophyllous scrub, and heath (202 and 189 species respectively) and different types of woodlands including mixed woodland (187 species), broad-leaved deciduous forests (186 species), coniferous woodland (156 species). Humid grasslands and tall herb communities comprise 171 species (Table 10.5).

The biotopes with the largest absolute numbers of species threatened across Europe are also mainly grasslands: mesophile grasslands (39 threatened species), dry calcareous grasslands and steppes (37), alpine and subalpine grasslands (34) and humid grasslands and tall herb communities and dry siliceous grasslands (27). Different types of woodlands generally hold lower numbers of threatened butterflies: mixed woodland (29 threatened species), broad-leaved deciduous forests (25) and coniferous woodland (23) while heath and scrub have 25 species considered threatened.
In contrast, the biotopes supporting the greatest proportion of threatened species are dominated by bogs and marshes (including blanket bogs, raised bogs, fens, transition mires and springs, water-fringe vegetation), humid grasslands and tall herb communities. These are followed in importance by mesophile grasslands and different types of woodlands (mixed woodland, coniferous woodland, broad-leaved deciduous forests), and different types of dry grasslands (dry calcareous grasslands and steppes, alpine and subalpine grasslands, dry siliceous grasslands). Note that specific biotopes such as volcanic features, islets and rock stacks or inland rocks, screes and sands are not discussed owing to their low total number of species mentioned, although they can be of great importance locally (e.g. volcanic features with *Hipparchia maderensis* on Madeira and *Scolitantides orion* in Eastern Europe, islets and rock stacks with *Parnassius apollo* or inland rocks, screes and sands with *Glaucopsyche alexis*).

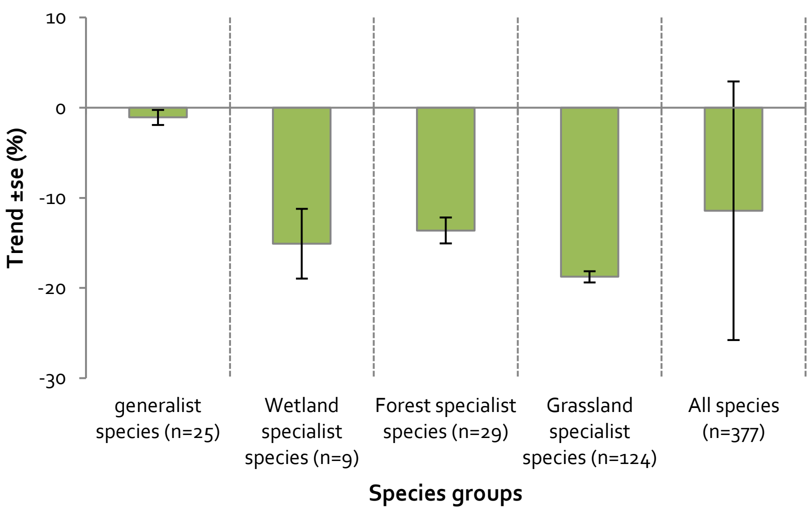


Figure 10.1: European trends of selected species groups according to broad biotope classes and specialism.

European trends for specialists and generalists

Our overall results show that butterflies are declining substantially across Europe, with a decline in distribution of -11% over the last 25 years (Figure 10.1). The results also show that the 25 most generalist species, according to our ranking (see above), did not significantly decline (-1%, t = -0.4809, p=0.63) compared to specialist butterflies. The biggest declines in distribution are among grassland specialists (-19%, t = -8.0075, p<0.001), followed by wetland species (-15%, t = -4.8188, p<0.01), and forest species (-14%, t = -1.9428, p<0.05).

Threats

Data on suspected threats were gathered only for the 71 species classed as threatened at the European level and are shown in Table 10.6. The majority of species (n = 63, almost 90%) are affected by agricultural improvement, which includes a wide range of activities from conversion of unimproved grasslands to arable crops, through to fertilisation of pastures. Although affecting only 33 species, land drainage is the major threat for wet grassland and wetland butterflies. Drainage immediately destroys the biotope of these butterflies, and is mostly followed by agricultural improvements. Characteristic species that suffer heavily from drainage are *Coenonympha oedippus* and *C. tullia*.

Table 10.6: Average grade of threat for threatened butterflies in Europe as well as per main biotope group, with the highest threatgrade per biotope group in bold.
N=total number of species.
Average grade of threat: 1=low, 2=medium, 3=high.
****=mentioned for less than three species.***

Threat	All biotopes	Forest	Grassland	Wetland	N
Land drainage	**2.2**	1.7	**2.2**	**2.4**	33
Agricultural improvements	2.1	1.9	**2.2**	2.0	**63**
Land claims / coastal development	2.1	2.0	2.1	*	41
Agricultural abandonment	2.1	1.9	**2.2**	1.9	46
Felling/destruction of woodland	2.1	**2.2**	2.0	1.7	45
Isolation and fragmentation of habitat	2.1	2.1	2.0	2.0	62
Afforestation on non-woodland habitats	1.9	1.8	1.9	2.0	53
Abandonment and change of woodland management	1.9	**2.2**	1.8	1.7	45
Recreational pressure and disturbance	1.8	1.9	1.8	2.0	48
Natural ecological change	1.8	2.0	1.7	*	37
Built development (inc. roads, housing, etc.)	1.8	1.8	1.8	1.7	58
Chemical pollution (inc. herbicides and pesticides)	1.8	1.6	1.8	1.6	55
Climatic change	1.7	2.1	1.6	1.6	45
Collecting (killing or taking)	1.4	1.5	1.4	1.5	46

Other important threats derive from the abandonment of agricultural land and changing biotope management. This is thought to affect 65% of the threatened species and is symptomatic of the widespread cessation of traditional farming systems that is known to have a negative impact on a variety of other wildlife groups (Poole et al. 1998; Tucker and Heath 1994). Examples of changing management include the cessation of cutting of damp hay meadows (affecting species like *Maculinea nausithous*, *M. teleius*, and *Lycaena helle*) and abandonment of pasture land (affecting species such as *Euphydryas aurinia* and *Maculinea alcon*).

The increasing use of herbicides and pesticides on farmland is also reported to be a serious problem for butterflies (affecting 80% of threatened species), especially in some eastern countries where economic pressures are more severe and regulations are less strict. Building developments such as roads, quarries and housing are also important (affecting 80% of threatened species). As a result of this massive direct loss of breeding areas, a growing threat arises from the subsequent isolation and fragmentation of biotopes which now affects 87% of threatened species.

Similar problems of abandonment and changing management were also reported in woodland biotopes, affecting 63% of threatened species. The main problem in woodlands seems to be loss of open woodland habitats following a shift from traditional management such as short-rotation coppice systems to high forest systems. This has been recognised as a major problem in western countries for many years (e.g., Warren and Key 1991) but there is growing evidence that this is a widespread and serious problem across Europe (e.g., Benes et al. 2002). The shift from traditional short-rotation standing crop to intensive high forests has a very negative impact on characteristic woodland butterflies as *Lopinga achine* (Bergman 2001). Afforestation of non-woodland biotopes is also a major threat to many species, especially those occurring in small breeding areas such as *Parnassius apollo*.

Discussion

Biotopes and their threats

This paper presents the first objective overview on the biotope requirements of almost all European butterflies as well as the chief threats to threatened species. Unlike preceding descriptions, the material has been collected in a standardized way over the whole of Europe, giving a unique insight into the threats for this insect group.

The results show that butterflies are highly dependent on man-made biotopes such as dry grassland and meadows, which are typically maintained by traditional forms of farming management such as livestock grazing and hay-making. A wide range of factors associated with the rapid intensification of agriculture across the region threatens such biotopes. Although dry grasslands are the richest in butterfly species, the most important biotopes for threatened butterflies are wet biotopes such as bogs and marshes. These are under particular threat from drainage, either to create fertile agricultural land or, in some cases, to control disease-bearing insects such as mosquitoes.

Contrary to many people's views of threats to butterflies, collecting was reported to be only a very minor or local importance. However, there were some important exceptions of species which are possibly quite seriously threatened by collecting, notably *Parnassius apollo, Polyommatus humedasae, Polyommatus poseidon, Polyommatus damone, Euphydryas maturna* and *Coenonympha oedippus*. Nevertheless, all these species are suffering far more seriously from problems such as biotope loss or changing biotope management.

Climatic change is also mentioned as a potential threat to several species, notably highly restricted montane endemics which are closely adapted to specific vulnerable biotopes and which have a very limited possibility of adapting to global warming (Dennis 1993; Wilson et al. 2005).

When considering threats, it is worth stressing that Europe is a large and diverse region, and it is therefore clear that the types of threat vary considerably from country to country. This partly reflects the fact that the types of biotope used by each species vary naturally across different climatic zones, but also reflects the wide variation of economic and political situations. Threats vary from site to site and have been examined further in the Prime Butterfly Areas of Europe report (van Swaay and Warren 2003). It is likely that most major threats identified for butterflies will continue to operate in the foreseeable future, and may even become more serious in some countries. For example, Eastern European countries have already started to suffer from serious agricultural intensification (e.g., Donald et al. 2001; Konvicka et al. 2006) and the problem may be exacerbated further now that their markets are becoming more open. The speed of change in some countries may also increase rapidly now they have joined the European Union and have access to extra subsidies for increased production. This poses a particularly serious potential threat as these countries hold a disproportionate large number of threatened butterflies.

On the plus side, there is a growing move to reform EU agricultural and forestry policies to encourage more environmentally sustainable systems, for example within mechanisms such as the Agri-environment Regulation (EU Reg. 2078/92). Although schemes currently being funded under such regulations comprise a very small proportion of the agricultural budget, they have the potential to slow down some of the trends reported. However, much wider reforms of agricultural policies are also urgently needed (e.g., see Tucker and Heath 1994; Baldock et al. 1994; Poole et al. 1998). Policies such as the EU Habitats and Species Directive may also help to slow declining trends but many countries have been slow to implement this Directive (e.g., Flanders - Maes and Van Dyck 2001) and its likely impact on butterflies remains uncertain.

Recent studies have shown that many montane species are shifting their distributions to higher altitudes, presumably as a result of climatic warming, and montane and boreal species may be threatened in future (Wilson et al. 2005).

Trends and comparison with other groups

The overall decline of butterflies at a European level confirms many previous observations (e.g., Heath 1980) and reflects the widespread loss of biodiversity reported in many other taxa (e.g., Delbeare 1998). However, for the first time we show that declines have been far more rapid in specialist species of grasslands, wetlands and forests. Our results show that butterflies seem to be reacting differently compared to a recent study describing biotope related trends in breeding birds (Gregory et al. 2005). Whereas our paper measured trends amongst specialists, the bird trends focused on communities (e.g., farmland birds and woodland birds). Although the methods of the two studies were different, the results make an interesting comparison.

While farmland birds (which occur in arable biotopes as well as managed grasslands), show an annual population decrease of –1.5% (from 1980 to 2002), grassland butterflies showed an annual distribution decrease of– 0.8% (for the 25 year period pre 1997). However, the rates of change cannot be compared directly because the butterfly trends are calculated from distribution data that substantially underestimate population decline (e.g., Thomas and Abery 1995; Warren et al. 1997).

In contrast, trends in woodland birds show little change compared to forest specialist butterflies, which showed an annual distribution decrease over this period of –0.01% to –0.6%. The comparatively rapid decline of forest butterflies suggests that they are more sensitive than birds to changes in this biotope. In woodlands, the decline of butterflies is probably linked with the loss of open woodland or forest clearings, whereas many of the birds studied are associated with closed forests where change has been less dramatic. It should also be noted that the butterflies studied have been pre-selected as specialists as opposed to woodland birds, which may occur in a range of other biotopes. Nevertheless the study supports the findings of Thomas et al. (2004) that butterflies are declining at least as fast as birds and possibly faster in many biotopes.

Butterflies are likely to respond to different factors than birds and, because of their annual life cycles, are likely to react more quickly (Thomas 1994). Butterflies tend to breed in smaller habitat patches and are more likely to reflect changes occurring at a finer scale. Thus, they provide additional and complementary information to birds, which tend to range more widely and have populations that operate over larger areas. Contrary to woodland birds, that can occur in dark forests, woodland butterflies are only found in open places, paths and glades where sun reaches the ground and nectaring flowers are found.

Conclusions

Our study demonstrates that data currently available for butterflies can be successfully used to produce generic trends at a continental scale as well as trends within different broad biotope types. The sensitivity of butterflies to environmental change and the availability of suitable data from many countries across Europe suggest that butterflies are very good candidates to build biodiversity indicators. Along with other major groups such as birds, they are therefore ideal candidates to monitor performance regarding the EU target to halt biodiversity loss by 2010. No equivalent data are available for other invertebrate taxa, making butterflies unique in enabling an assessment of trends in this exceptionally diverse and ecologically important group.

There is a growing network of specialist Lepidoptera groups in countries across Europe, many of them using volunteers to compile extensive datasets on butterflies and their trends. Datasets for butterflies include traditional mapping schemes to identify trends such as those used in the Red Data Book (Van Swaay and Warren 1999) but also detailed monitoring schemes based on weekly transect counts at networks of sites. A summary of the schemes currently in operation is given in the country summaries of the Prime Butterfly Areas of Europe (Van Swaay and Warren 2003). New monitoring schemes are being started or planned in other countries and the monitoring network is being developed each year. A new organization, called Butterfly Conservation Europe, has been started to co-ordinate and collate such data and to provide the support for volunteers and organizations who contribute (see www.bc-europe.eu). The infrastructure needed to obtain butterfly data at a European level is thus already well developed and, given sufficient resources, could produce an even more scientifically robust method of monitoring change in the future.

Appendix 10.1: Habitat profiles of European butterflies (listed in alphabetical order with taxonomy according to Karsholt & Razowsky, 1999). 1=5-10%; 2=10-20%; 3=20-30% etc.

		Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
Species	No of Countries	coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Aglais urticae	42									1	1	1		1													1	1	2	2	1
Anthocharis cardamines	43									2	2	2		2														1			
Anthocharis damone	4			4	4		4																								
Anthocharis euphenoides	4				3	1	2	2	1			1				1							1								
Anthocharis gruneri	5				3	3	3														3										
Apatura ilia	33											4		3	3				1												
Apatura iris	35											4		3	2														1		
Apatura metis	13											3		3	3				2												
Aphantopus hyperantus	38							1		2	2	2		1														1			
Aporia crataegi	39			1			1				1	2		2													1	2		1	
Araschnia levana	33									1	1	2		2	1												1	1	2		1
Archon apollinus	3			5	5																										
Arethusana arethusa	22						4	3			1	1										1									
Argynnis adippe	39						1	1		1	2	2	1	2																	
Argynnis aglaja	41						1	1	1	1	2	2		2																	
Argynnis elisa	2						5					5																			
Argynnis laodice	15									3	1	2	1	2	1			1	1	1								1			
Argynnis niobe	39			1			1	2	1	1	2	2		2																	
Argynnis pandora	23				1		2	1	1		1	3		1		1												1			
Argynnis paphia	42										1	3	2	3														1			
Aricia agestis	36						2	2			2	1																			1
Aricia anteros	10						4		4		2										2										
Aricia artaxerxes	31			1			2	1	2	1	2	1		1																	
Aricia cramera	4			2	2				2							2												2			2
Aricia eumedon	31						2		1	3	2	1							1	1											
Aricia morronensis	1							3	3												3	3									
Aricia nicias	9							1	2	3	2	1	2	2																	
Aricia teberdinus	2								9																						
Boloria alaskensis	3								4	4	4																				
Boloria angarensis	3								3	3			3				3														
Boloria aquilonaris	23									2			1				4	2	1	2											
Boloria chariclea	6			2	2				2	2	2																				
Boloria dia	34						2	1		1	2	1		1	1																
Boloria distincta	2																					9									
Boloria eunomia	23								1	3							3	1		2											
Boloria euphrosyne	40							1		2	2	2	2	2			1														
Boloria freija	10			1					2	1	1		1				5			1											
Boloria frigga	10								1	1	1						7			1											
Boloria graeca	10								5				4	3																	
Boloria improba	6			2	2				2	2	2																				
Boloria napaea	10			2	2				7									2													
Boloria pales	17			2					8									1													
Boloria polaris	6			2					2	2	2											2									
Boloria selene	34							1		2	2	1	1	1																	
Boloria selenis	5								2	2	5	2		2																	
Boloria thore	14								3	2			1	1						2											
Boloria titania	21								2	2	2	1	1	2																	
Borbo borbonica	1				5																										

		Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
Species	No of Countries	coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Brenthis daphne	27			1			1	1		1	2	2		2														1			
Brenthis hecate	21						2	2	1		3	2		1																	
Brenthis ino	36									4	2	1		1					1												
Brintesia circe	26				1		2				1	3	1	2								1									
Cacyreus marshalli	1																												5	5	
Callophrys avis	3			3	5											3															
Callophrys butleri	2			3			3	3														3									
Callophrys rubi	41			2			1					2	1	2			1											1	1		
Carcharodus alceae	31						3	2			2	1															1	1			2
Carcharodus baeticus	5	4						4													4										
Carcharodus floccifera	29						2	1		1	3	2		1																	
Carcharodus flocciferi	1										5	5																			
Carcharodus lavatherae	23				1		4	2				2																			
Carcharodus orientalis	10				1	2	5	1	1		2																				
Carcharodus stauderi	2		5			5																									
Carterocephalus palaemon	34									2	2	3	1	2																	
Carterocephalus silvicola	15											3	2	4	2																
Celastrina argiolus	42			1								3		2	1													2	1	1	
Charaxes jasius	10	1			4	3	3	1								1															
Chazara briseis	27						4	2				2										2									
Chazara persephone	4						2	4	2					2																	
Chazara prieuri	2				5		5																								
Chilades trochylus	4				5	5																									
Coenonympha amaryllis	2						5	5																							
Coenonympha arcania	37			1			2	1			2	2		2														1			
Coenonympha corinna	1				5		5																								
Coenonympha darwiniana	2								9																						
Coenonympha dorus	4			2	2		5	2																							
Coenonympha gardetta	9								6		3	3																			
Coenonympha glycerion	31						2	1		2	3	1		2														1			
Coenonympha hero	21									3	2	2		2						1											
Coenonympha leander	9						1	3	1		3	3		3																	
Coenonympha oedippus	14									3	1	1		1			2	2	1	1											
Coenonympha pamphilus	42						2	2		1	2														1				1		1
Coenonympha rhodopensis	6						3		5		4																				
Coenonympha thyrsis	1				5		5																								
Coenonympha tullia	30								1	2	1						3	3		2											
Colias alfacariensis	25			1			3	2			1														1	1					1
Colias aurorina	4				2		2	2	2				2																		
Colias caucasica	6			2					4				4																		
Colias chrysotheme	10					1	5	4			1																				1
Colias croceus	26						1				1														1	1		1			1
Colias erate	15				1		3	1			1														1	2					1
Colias hecla	5	2		2					5	2	2																				
Colias hyale	27						2	1			2														2	1					2
Colias myrmidone	17						3	3			3		1	1																	
Colias nastes	5			2			2		4	2	2																				
Colias palaeno	21								2	1			2	1			4	2		1											
Colias phicomone	8								9	2																					
Colotis evagore	1						5																								5
Cupido alcetas	21			1			2	1	1	2	2	2	1	2	1																
Cupido argiades	33						1	1		2	2	1			1										1						1

		Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
Species	No of Countries	coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Cupido decolorata	16				1		2	2			2	2			2																
Cupido lorquinii	2			5	5																										
Cupido minimus	41						2	2	1	1	2	1		1																	
Cupido osiris	20						3	1	2	1	2		1	1																	
Danaus chrysippus	3				2	2																			2						
Danaus plexippus	3																											2			2
Erebia aethiopella	1								4			4				4															
Erebia aethiops	29						1		1		2	2	3	3																	
Erebia alberganus	6								4		4	4																			
Erebia calcaria	3								5												3	3									
Erebia cassioides	12								6												3	2									
Erebia claudina	1								9																						
Erebia cyclopius	4									3			5	3																	
Erebia dabanensis	2																					9									
Erebia disa	6								3	2				2			4			2											
Erebia discoidalis	3								4	4							4														
Erebia embla	9									1			3	1			5			1											
Erebia epiphron	20			2				1	7																						
Erebia epistygne	2						5		5																						
Erebia eriphyle	4								9																						
Erebia euryale	23			1					4			1	3	2																	
Erebia fasciata	4								4	4				4																	
Erebia flavofasciata	2								9																						
Erebia gorge	17								5												4	1									
Erebia gorgone	3								7												4										
Erebia graucasica	2							5	5																						
Erebia hispania	3								7												4										
Erebia iranica	2									9																					
Erebia lefebvrei	3								4												7										
Erebia ligea	30								1			2	3	3																	
Erebia manto	13			2				1	7				1			1															
Erebia medusa	25						2	1	2	1	2	1	2	2																	
Erebia melampus	6						2		7		2																				
Erebia melancholica	2								5	5																					
Erebia melas	11								3		1		3								2	1									
Erebia meolans	8			1					6		2	1	1																		
Erebia mnestra	3								9																						
Erebia montana	5								3												3	5									
Erebia neoridas	3										4		4								4										
Erebia nivalis	3								4												4	4									
Erebia oeme	15			2					7	1	2																				
Erebia orientalis	3								8					3																	
Erebia ottomana	9						2	2	3		2		4								2										
Erebia palarica	2								5		5																				
Erebia pandrose	21			2					7																						
Erebia pharte	9								9													2									
Erebia pluto	6								4												4	2									
Erebia polaris	6							4			4																				4
Erebia pronoe	16			2					5												1	2									
Erebia rhodopensis	5								4		2		4																		
Erebia rossii	3								5		5																				
Erebia sthennyo	2								5	3												3									

		Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
Species	No of Countries	coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Erebia stirius	5								4												2	4									
Erebia styx	5								4												4	4									
Erebia sudetica	5								5		2		2	2								2									
Erebia triaria	10								6		3	2	2																		
Erebia tyndarus	6								5							2					2	3									
Erebia zapateri	2												9																		
Erynnis marloyi	7				2	2	4	2													2										
Erynnis tages	40			1			2	2			2	1																			
Esperarge climene	9			1	1		1					4		2		1															
Euchloe ausonia	19						4	1	2		1																	2			2
Euchloe belemia	3				3		3	3																							3
Euchloe charlonia	2						9																								
Euchloe crameri	3				2	1	2	2	1	1	1												1		1	1		2			2
Euchloe insularis	1				2	2	2	2	2																						
Euchloe penia	5				3	3	3														3										
Euchloe simplonia	2				2	2	2	2	2	2	2												2								
Euchloe tagis	3				2	2	3	2								2							2								2
Euphydryas aurinia	39			1			1	1	1	3	3	1																			
Euphydryas cynthia	7								3	1	1	1	1	1	1	1	1	1	1	1	1	1									
Euphydryas desfontainii	3			3	3		3									3															
Euphydryas iduna	6			2	2				3	2							3														
Euphydryas intermedia	8			1					1	1	2	1	1	1	1	1	1	1	1	1	1										
Euphydryas maturna	27									2	2	4		2	1																
Gegenes nostradamus	11				4	4	3																								
Gegenes pumilio	9				3	3	3															2									
Glaucopsyche alexis	38				1		2	2			2	2																			1
Glaucopsyche melanops	3			4	4							2																			
Gonepteryx cleopatra	14			1	3	1										2															
Gonepteryx farinosa	7				3							5									3										
Gonepteryx maderensis	1										5					5															
Gonepteryx rhamni	41			1								2	1	2														1	1		
Hamearis lucina	34			1			2			2	2	3		2																	
Hesperia comma	40			1			2	2	1		2																				1
Heteropterus morpheus	28			1						3	2	1		1	1				1												
Hipparchia alcyone	20			1	1		2	2				2	4	1									1								
Hipparchia aristaeus	4				2	2	2							2							2										
Hipparchia autonoe	2						5	5																							
Hipparchia azorina	1			4							4																				
Hipparchia christenseni	1				5	5																									
Hipparchia cretica	1				5	5																									
Hipparchia fagi	22			1			1	2				3	1	1								1									
Hipparchia fatua	7				2		5						2	2							2										
Hipparchia fidia	3			4	4		2									2															
Hipparchia maderensis	1							2					2	2		2								2							
Hipparchia mersina	2												9																		
Hipparchia miguelensis	1			4							4																				
Hipparchia neomiris	1			5			5																								
Hipparchia occidentalis	1			4							4																				
Hipparchia pellucida	5				1	2	1	1	2				1	1							1										
Hipparchia semele	33	2		2			2	2				1	2	1								1	1								
Hipparchia statilinus	28			1			2	3				1	1									1	1								
Hipparchia syriaca	10						5					2	2	2		2															

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Hipparchia volgensis	7				1	1		3	1			2		2								2									
Hyponephele huebneri	1							9																							
Hyponephele lupinus	20				1	1	2	2				2		1																	
Hyponephele lycaon	32			1			3	3			1	1	1																		
Inachis io	42									1	1	1		1														1	2	2	1
Iolana iolas	15				2	1	2	1		2	2																				
Iphiclides podalirius	31			1	1		1				1	2		1													2	1	1		
Issoria eugenia	2								7				4																		
Issoria lathonia	37			1			2	2			2	1													1	1		1			2
Kirinia roxelana	11				3							6				2															
Laeosopis roboris	4											2			4				2									4			
Lampides boeticus	23			1	2	2	1	1				1														1	1	1			1
Lasiomata megera	1			3			3																				3	3			
Lasiommata deidamia	3									4			4	2								2									
Lasiommata maera	37			1			1				2	2	1	2																	
Lasiommata megera	39			1			2	2			1	1										1						1			1
Lasiommata paramegaera	1				5			5																							
Lasiommata petropolitana	29			1					2		1	1	3	3																	
Leptidea duponcheli	10		1	1	1	1	3	3		1				1														1			
Leptidea morsei	15									2	2	4	2	2																	
Leptidea sinapis complex	41						1	1		1	2	2	1	1														1			
Leptotes pirithous	19				2	2	2	2																				1	1		
Libythea celtis	19						2					2		2														1	2		
Limenitis camilla	37											6		3																	
Limenitis populi	31						1	1				4		3															1		
Limenitis reducta	25				1		1					5	1	2																	
Lopinga achine	28											4	2	4	1																
Lycaena alciphron	31			1			2	2	1	2	2	1																			
Lycaena candens	7								5		5																				
Lycaena dispar	34						1			3	2	1						1	2	1								1			
Lycaena helle	23								1	4	1								1	1											
Lycaena hippothoe	33								2	4	3									1											
Lycaena ottomanus	8				3	3	3				2			1																	
Lycaena phlaeas	44			1			2	2	1		2	1																			
Lycaena thersamon	20						3	2			1	1																	1		1
Lycaena thetis	3					4	4					4																			
Lycaena tityrus	34			1			1	2	1	1	2	1														1					
Lycaena virgaureae	35						1	1	1	2	3	1	1	2																	
Maculinea alcon	28			1			1			3	2	1								1											
Maculinea arion	39			1			2	2	1	1	2	1	1																		
Maculinea nausithous	21							1		4	2							2	2	1											
Maculinea rebeli	15						5	2	4		1																				
Maculinea teleius	21									4	3							2	1	1											
Maniola chia	1				9																										
Maniola halicarnassus	2				5		5																								
Maniola jurtina	42						1	1		1	2	1		1														1	1		1
Maniola megala	2																		5	5											
Maniola telmessia	2				5		5																								
Melanargia galathea	29						2	2		1	3	1		1														1			
Melanargia ines	2				5		5																								
Melanargia lachesis	6				3		3				2					2					2							2			
Melanargia larissa	8				2	2	4															4									

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Melanargia occitanica	3				4		4				2																				
Melanargia russiae	13				1	1	3	3	1	1	2		1	1																	1
Melitaea aetherie	2															9															
Melitaea arduinna	8						2	2	1		2	2		1																	
Melitaea asteria	2								9																						
Melitaea athalia	41			1			1	2		1	2	2	1	1																	
Melitaea aurelia	27			1			3	2	1	2	3																				
Melitaea britomartis	16						3	2		1	3	1																			
Melitaea cinxia	40						2	2		1	3																				
Melitaea deione	5						3	3	2		3																				
Melitaea diamina	36						1	1		4	2									1											
Melitaea didyma	32						3	3			1	1																			
Melitaea parthenoides	7						2	2	3		3				1					1											
Melitaea phoebe	34						2	2		1	3	1																			
Melitaea trivia	20						3	2		2	4	2																			
Melitaea varia	3								9																						
Minois dryas	28						2	1		1	2	2			1																
Muschampia cribrellum	5					2	4	3																							1
Muschampia proto	11			2	3	3	3	1	2																						
Muschampia tessellum	10			1		1	3	2			2																				
Neolycaena rhymnus	3			2			2	5													2										
Neozephyrus quercus	42											6		2		1												1	1		
Neptis rivularis	20										1	4	1	2	2													1			
Neptis sappho	19											5		3	2																
Nymphalis antiopa	39											3	2	2	1													1	1		
Nymphalis polychloros	39											2		2													1	2	2		
Nymphalis vaualbum	17											4	1	2	1													1	1	1	
Nymphalis xanthomelas	24											3	1	3	2												1	1	1	1	
Ochlodes venata	39							1		2	2	1		1														1			
Oeneis bore	5			2					2	2											2	2									
Oeneis glacialis	4								5										2		2	2									
Oeneis jutta	11								1	1			3				7														
Oeneis melissa	2								4	4												4									
Oeneis norna	5			2					2	2	2						4														
Oeneis patrushevae	2									5							5														
Oeneis polixenes	1								5	5																					
Oeneis tarpeia	3						4	4	2																						
Papilio alexanor	8			2	2	4	1	1																							
Papilio hospiton	1			2	2	2	2	2	2																						
Papilio machaon	43						1	1		1	2	1															1		1	1	1
Pararge aegeria	45											3	2	2	1													1	1		
Pararge xiphia	1															9															
Parnassius apollo	30						2		3		1	1	1								1	2									
Parnassius mnemosyne	34						1		2	2	2	3		1																	
Parnassius phoebus	7								5	2	2								2	2		2									
Pieris balcana	6											7	2	2																	
Pieris brassicae	44									1	1														1	2	1	1	2	2	1
Pieris bryoniae	15			1					3	1		1	2	1																	
Pieris ergane	14				1		3	1				2		1							2	2				1					
Pieris krueperi	5						3														3	3				3					
Pieris mannii	18				1		2	2				2		1								1									
Pieris napi	42									1	1	1		1												1	1	1		1	

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Pieris rapae	44										1														1	1	1	1	2	1	1
Plebeius argus	41			2			2	2			2		1	1																	
Plebeius argyrognomon	30						3	2		1	2	1													1	1					
Plebeius eurypilus	2				5	5																									
Plebeius glandon	12			1			1	1	7												2										
Plebeius hesperica	1						5																								5
Plebeius idas	37			2			1	2	1		2	1		2																	
Plebeius loewii	2				5	5																									
Plebeius optilete	24			1					1				3				4														
Plebeius orbitulus	9								8												2	2									
Plebeius psylorita	1				5	5																									
Plebeius pylaon	11				1	1	5	2			1											2									
Plebeius pyrenaica	8								4	2	3										2										
Plebeius sephirus	1						9																								
Polygonia c-album	41											2		2													1	2	2	1	
Polygonia egea	12				3	3	3	1																							
Polyommatus admetus	13						5		1		2	4																			
Polyommatus albicans	1				5		5																								
Polyommatus amandus	34						2	2		1	3	1													1						
Polyommatus andronicus	1								5		5																				
Polyommatus aroaniensis	2				4		4														4										
Polyommatus bellargus	32						4	2	1		2	1																			
Polyommatus caucasica	1						3		3	3	3																				
Polyommatus coelestina	5						8				3																				
Polyommatus coridon	31			1			4	2			2	1																			
Polyommatus cyane	2						9																								
Polyommatus damocles	2						5	3														3									
Polyommatus damon	23						4	2	2		1	2																			
Polyommatus damone	3						5	3			3																				
Polyommatus daphnis	27						4	2			2	1		1																	
Polyommatus dolus	1				4		4	4																							
Polyommatus dolus (ainse)	1				5		5																								
Polyommatus dorylas	29						4	2	2		2	2																			
Polyommatus eroides	14						2	3	2	1	2		1	1									1								
Polyommatus eros	10						4		6												2										
Polyommatus escheri	13				1	2	4	1	1	2	1										1	1									
Polyommatus fabressei	1				5		5																								
Polyommatus fulgens	1						9																								
Polyommatus golgus	1								5												5										
Polyommatus hispana	2			2	2		4		2																						
Polyommatus icarus	44						2	2		1	2																		1		1
Polyommatus iphigenia	2				5		5																								
Polyommatus kamtschadalus	1							5	5																						
Polyommatus menelaos	1						5				5																				
Polyommatus nephohiptameno	2						4				4										4										
Polyommatus nivescens	1				4		4															4									
Polyommatus philippi	1						5				5																				
Polyommatus poseidon	2						4		4													4									
Polyommatus ripartii	13				2		6	2	1		1			1																	
Polyommatus semiargus	41						1	1	1	2	3	1		1																	1
Polyommatus thersites	25			1	1		3	2	1		2	1																			1
Pontia callidice	12								7	1				1							1		1		1						1

Species	No of Countries	Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
		coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Pontia chloridice	12						3	2	1		2										2	2									2
Pontia daplidice	31						1	1			1	1													1	1		1	1		2
Proterebia afra	6			2	2	3	2	3	2					2																	
Proterebia afra krymea	1								9																						
Pseudochazara alpina	1							9																							
Pseudochazara amymone	1				9																										
Pseudochazara anthelea	7				2	2	4														2										
Pseudochazara cingovskii	2						9																								
Pseudochazara euxina	2								5													5									
Pseudochazara geyeri	5						7	4																							
Pseudochazara graeca	2						7															4									
Pseudochazara hippolyte	3			2			4	4													2										
Pseudochazara mniszechii	2				5								5																		
Pseudochazara orestes	2				4	4															4										
Pseudochazara quirensis	1							9																							
Pseudophilotes abencerragu	2						5																								5
Pseudophilotes baton	13			2	2	1	4	3	1		1																				
Pseudophilotes bavius	9				1	2	5	2													1										
Pseudophilotes vicrama	23			1	1	1	3	3					1																		
Pyrgus alveus	33			1			2	2	2		3																				
Pyrgus andromedae	16								6													1									
Pyrgus armoricanus	26			1			4	2	1	1	2																				
Pyrgus bellieri	3								4	4	4																				
Pyrgus cacaliae	10						1		8	1				1																	
Pyrgus carlinae	2						2	2	2	2	2												2								
Pyrgus carthami	29						3	2	1	1	2	1									1										
Pyrgus centaureae	5								2				2				6	2													
Pyrgus cinarae	9		1	1			3	2		1											1	1									
Pyrgus cirsii	9						2	2	2		2												1								1
Pyrgus malvae	40			1			2	1		2	2	1		1																	
Pyrgus malvoides	7			1	1	1	3	2	1	1	2														1						
Pyrgus onopordi	6				2		4	2	2	2	2														2						
Pyrgus serratulae	32						3	3	2		2																				
Pyrgus sidae	18				1	1	3	1			1		1								1										
Pyrgus warrenensis	4								9																						
Pyronia bathseba	3				2	2	2	2																							
Pyronia cecilia	8				4	2	3	2								2															
Pyronia tithonus	26			1			2	1		1	2	2			1													2			
Satyrium acaciae	26			3			2				1	4		2																	
Satyrium esculi	4				1							1				2											1	2	1	1	1
Satyrium ilicis	36			2								5		2																	
Satyrium ledereri	2				5	5																									
Satyrium pruni	34			1								3		2													1	2	1		
Satyrium spini	31			2			2	1				3		2														2			
Satyrium w-album	39											4		3														2	2		
Satyrus actaea	6			2			5	2	2		2																				
Satyrus ferula	16					1	2	2	2			2			2						2										
Scolitantides orion	29						3	2	1			1									1	2									
Spialia orbifer	15						3	2			3				2																
Spialia phlomidis	7				3	1	3	2													1										
Spialia sertorius	20			1			4	2		1	1																				
Spialia therapne	1				4	4	4																								

		Coastal		Heath and scrub			Grassland					Forest					Wetland				Unvegetated				Agriculture				Urban		
Species	No of Countries	coastal sand-dunes and sand beaches	cliffs and rocky shores	heath and scrub	sclerophyllous scrub	phrygana	dry calcareous grasslands and steppes	dry siliceous grasslands	alpine and subalpine grasslands	humid grasslands and tall herb communities	mesophile grasslands	broad-leaved deciduous forests	coniferous woodland	mixed woodland	alluvial and very wet forests and brush	broad-leaved evergreen woodland	raised bogs	blanket bogs	water-fringe vegetation	fens, transition mires and springs	screes	inland cliffs and exposed rocks	inland sand-dunes	volcanic features	improved grasslands	crops	orchards, groves and tree plantations	tree lines, hedges, small woods, bocage, parkland dehesa	urban parks and large gardens	towns, villages, industrial sites	fallow land, waste places
Tarucus balkanica	10				6	5																									
Tarucus theophrastus	1			5	5																										
Thecla betulae	41			1								3		2													1	2	2		
Thymelicus acteon	29				1		4	2			2	2																			
Thymelicus hyrax	2				5	5																									
Thymelicus lineola	40						1	2		1	2	1																1			2
Thymelicus sylvestris	37						2	2		1	3	2		1														1			
Tomares ballus	3				2		4	4																							2
Tomares callimachus	4						3	3				2		2							2										
Tomares nogelii	5					2	7					2																			
Tongeia fischeri	1						5	5																							
Triphysa phryne	4			2			4	4						2																	
Turanana endymion	2				9																										
Vanessa atalanta	22									1	1	1		1														1	1	1	
Vanessa cardui	15				1				1	1	2															1			1		1
Vanessa indica	1																														
Vanessa virginiensis	1																														
Ypthima asterope	3					9																									
Zegris eupheme	5						4	2											1						1						3
Zegris pyrothoe	2						7	4																							
Zerynthia cerisy	11			1	2	1	4				1			1																	
Zerynthia cretica	1			5	5																										
Zerynthia polyxena	22			1	1		2	1		2	2	1		1	1																
Zerynthia rumina	4	1		1	2	1	2	2	1	1	1	1											1					1			
Zizeeria knysna	3						3	3																					3		3

Appendix 10.2

List of specialist butterflies per biotope.

Forests

Apatura ilia, Apatura iris, Apatura metis, Argynnis paphia, Carterocephalus silvicola, Erebia aethiops, Erebia ligea, Esperarge climene, Euphydryas maturna, Gonepteryx farinosa, Hipparchia alcyone, Kirinia roxelana, Lasiommata petropolitana, Leptidea morsei, Limenitis camilla, Limenitis populi, Limenitis reducta, Lopinga achine, Neozephyrus quercus, Neptis rivularis, Neptis sappho, Nymphalis antiopa, Nymphalis vaualbum, Nymphalis xanthomelas, Pararge aegeria, Pieris balcana, Satyrium ilicis, Satyrium pruni, Satyrium w-album

Grassland

Arethusana arethusa, Aricia anteros, Aricia artaxerxes, Aricia nicias, Boloria graeca, Boloria napaea, Boloria pales, Boloria polaris, Boloria titania, Brenthis hecate, Brenthis ino, Carcharodus lavatherae, Carcharodus orientalis, Coenonympha dorus, Coenonympha gardetta, Coenonympha glycerion, Coenonympha leander, Coenonympha rhodopensis, Colias alfacariensis, Colias aurorina, Colias chrysotheme, Colias erate, Colias hecla, Colias myrmidone, Colias nastes, Colias phicomone, Cupido minimus, Cupido osiris, Erebia alberganus, Erebia cassioides, Erebia epiphron, Erebia eriphyle, Erebia gorge, Erebia manto, Erebia medusa, Erebia melampus, Erebia meolans, Erebia oeme, Erebia orientalis, Erebia pandrose, Erebia pharte, Erebia pronoe, Erebia sudetica, Erebia triaria, Erebia tyndarus, Erynnis marloyi, Erynnis tages, Euchloe ausonia, Euphydryas aurinia, Euphydryas cynthia, Glaucopsyche alexis, Hipparchia syriaca, Leptidea duponcheli, Lycaena alciphron, Lycaena candens, Lycaena helle, Lycaena hippothoe, Lycaena ottomanus, Maculinea arion, Maculinea nausithous, Maculinea rebeli, Maculinea teleius, Melanargia galathea, Melanargia russiae, Melitaea arduinna, Melitaea aurelia, Melitaea britomartis, Melitaea cinxia, Melitaea deione, Melitaea diamina, Melitaea didyma, Melitaea parthenoides, Melitaea phoebe, Melitaea trivia, Muschampia cribrellum, Neolycaena rhymnus, Oeneis glacialis, Parnassius mnemosyne, Parnassius phoebus, Plebeius argyrognomon, Plebeius glandon, Plebeius orbitulus, Plebeius pylaon, Plebeius pyrenaica, Polyommatus admetus, Polyommatus amandus, Polyommatus bellargus, Polyommatus coelestina, Polyommatus coridon, Polyommatus damon, Polyommatus damone, Polyommatus daphnis, Polyommatus dorylas, Polyommatus eroides, Polyommatus eros, Polyommatus escheri, Polyommatus ripartii, Polyommatus semiargus, Polyommatus thersites, Pontia callidice, Pontia chloridice, Pseudochazara anthelea, Pseudochazara geyeri, Pseudophilotes baton, Pseudophilotes bavius, Pseudophilotes vicrama, Pyrgus alveus, Pyrgus andromedae, Pyrgus armoricanus, Pyrgus cacaliae, Pyrgus carthami, Pyrgus cinarae, Pyrgus cirsii, Pyrgus malvoides, Pyrgus onopordi, Pyrgus serratulae, Satyrus actaea, Spialia orbifer, Spialia sertorius, Thymelicus acteon, Tomares callimachus, Tomares nogelii, Triphysa phryne, Zerynthia cerisy

Wetlands

Boloria aquilonaris, Boloria freija, Boloria frigga, Coenonympha tullia, Colias palaeno, Erebia disa, Erebia embla, Oeneis jutta, Pyrgus centaureae

11. Prime Butterfly Areas of Europe: An initial selection of priority sites for conservation

*Slightly modified from: Van Swaay, C.A.M. & Warren, M.S. (2006). Journal of Insect Conservation **10 (1)**, 5-11.*

Abstract

The Red Data Book of European Butterflies, published in 1999, showed that butterflies have declined seriously across Europe and that 71 of the 576 species are threatened (12% of the total) either because of their extreme rarity or rapid decline. They comprise 19 globally threatened species and 52 species threatened at a European level. Many more species were shown to be declining in substantial parts of their range and a further 43 species were classified as Lower Risk (near threatened).

A follow up project was conducted in 2002-3 to identify Prime Butterfly Areas in Europe where conservation should be targeted as a priority. Due to constraints of time and resources, the review in this chapter could not be comprehensive, and concentrated on identifying the most important (prime) areas for 34 target species, using a network of national compilers. The book gives details of 431 areas covering 1.8% of the land surface of Europe, and shows that target butterflies are declining in one quarter of PBAs, indicating that breeding habitats are continuing to deteriorate even though many are protected by national designation. Chief threats are from agricultural intensification, afforestation, abandonment of traditional practices, and isolation. The results of these two projects provide useful models of what can be achieved at a European scale and demonstrate the effective collaboration of country experts to achieve shared conservation objectives.

Humid grasslands in the Moerputten, one of the Prime Butterfly Areas in the Netherlands, selected because of the occurrence of Phengaris teleius.

Introduction

The decline of Europe's butterflies has been recognised for many years (e.g. Heath, 1981; Pavlicek-Van Beek et al., 1992; Pullin, 1995), but the full scale of the problem was not known until the publication of the Red Data Book of European Butterflies (Van Swaay & Warren, 1999).

The analysis showed that a total of 71 European species are threatened (12% of the total), comprising 19 that are threatened at a global level and 52 threatened at a European level. Amongst the globally threatened species (endemic to Europe):

- 1 species is Critically Endangered;
- 4 species are Endangered;
- 14 species are Vulnerable.

The European threat status (for species also found outside Europe) was:

- 1 species is Extinct;
- 6 species are Critically Endangered;
- 14 species are Endangered;
- 31 species are Vulnerable.

A further 43 species are classed as Lower Risk (near threatened).

In this paper we present a summary of a follow up project to identify Prime Butterfly Areas where conservation efforts should be focused. The project was one of several aimed at identifying Important Biodiversity Areas across Europe, which so far includes birds (Heath and Evans, 2000), and work in progress on plants (Anderson, 2002), reptiles and dragonflies.

The results are intended to support other initiatives, like Natura 2000, the Pan-European Ecological Network (PEEN), the Pan-European Biological and Landscape Diversity Strategy and the Bern Convention. Protection and proper management of these areas will not only help to conserve these species, but also many other characteristic butterflies and other invertebrates occurring in the same habitats.

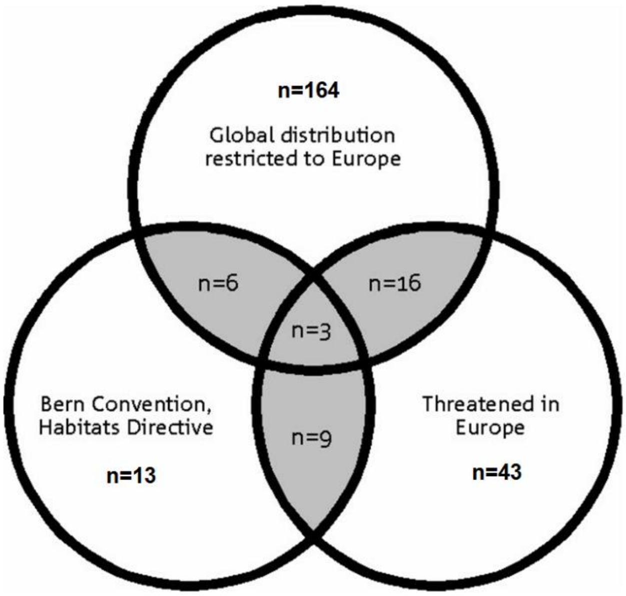

Figure 11.1. Selection of target-species used for the selection of Prime Butterfly Areas. Target-species fulfil at least two of three criteria (grey-shaded).

Methods

The principal aims of the project were:

- to identify an initial selection of the most important areas for the conservation of priority butterflies in Europe;
- to promote awareness of threatened butterflies in Europe, their special refuges and the main issues affecting them;
- to present information on each area in a standardised way;
- to help focus conservation and management measures on these areas.

Information was gathered from all European countries belonging to the Council of Europe, including Madeira, the Azores, the Canary Islands, Cyprus, the whole of Turkey and Russia east to the Urals. The selection of Prime Butterfly Areas (PBAs) was focussed on butterfly species fulfilling at least two of the following three criteria (grey-shaded in Figure 11.1):

1. Zoo-geography: the world range of the species is restricted to Europe (Range Affinity 4 in Van Swaay & Warren, 1999) (189 species).
2. Conservation: the species is threatened according to the Red Data Book of European Butterflies (Van Swaay & Warren, 1999) or the IUCN Red List of threatened animals (71 species).
3. Legislation: the species is listed in Appendix II of the Bern Convention (on the conservation of European wildlife and natural habitats) and/or the EU Habitats and Species Directive (23 species).

The 34 target-species selected by these criteria for inclusion in the Prime Butterfly Areas in Europe are listed in Table 11.1. A site is called a Prime Butterfly Area if it contains a substantial resident population of at least one of these target species. We included two types of areas: 1) discrete sites that support one or more rare or threatened; or 2) wider areas (such as mountain ranges or valley systems) where a target species occurs as scattered populations that may well be connected as a single metapopulation.
Furthermore information of each PBA was collected on location, protection status, trend and threats.

Euphydryas aurinia *is one of the target species for Prime ButterflyAreas.*

Table 11.1: List of target-species for Prime Butterfly Areas project, each of which fulfilled at least two of three criteria (grey-shaded in figure 11.1). For more details on the global distribution see Van Swaay & Warren (1999). Threatened species are listed as such by in the Red Data Book of European Butterflies or on the IUCN Red List of threatened animals.

Species	Global distribution restricted to Europe	Threatened	Bern Convention / Habitats Directive
Zerynthia caucasica	X	X	
Parnassius apollo		X	X
Papilio hospiton	X		X
Pieris wollastoni	X	X	
Pieris cheiranthi	X	X	
Gonepteryx maderensis	X	X	
Lycaena ottomanus	X	X	
Maculinea arion		X	X
Maculinea teleius		X	X
Maculinea nausithous		X	X
Maculinea rebeli	X	X	
Plebeius trappi	X	X	
Plebeius hesperica	X	X	
Polyommatus golgus	X		X
Polyommatus humedasae	X	X	X
Polyommatus galloi	X		X
Polyommatus dama	X	X	
Argynnis elisa	X		X
Euphydryas maturna		X	X
Euphydryas aurinia		X	X
Lopinga achine		X	X
Coenonympha oedippus		X	X
Coenonympha hero		X	X
Erebia christi	X	X	X
Erebia sudetica	X	X	X
Erebia epistygne	X	X	
Erebia calcaria	X		X
Melanargia arge	X		X
Hipparchia maderensis	X	X	
Hipparchia azorina	X	X	
Hipparchia occidentalis	X	X	
Hipparchia miguelensis	X	X	
Pseudochazara euxina	X	X	

Within the short time and limited funding available for this project, it was only possible to identify a first selection of the most important areas for target species in Europe, combined with a wide geographic coverage that includes both marginal and core populations. In general, we aimed to include the three best populations of each target species within each country. As with the Red Data Book, the data were provided by over 50 national compilers who were asked to select the Prime Butterfly Areas for their country and complete a questionnaire giving details on location, key butterfly species, habitats and land uses, threats, protection, and conservation issues (following the classification of Tucker and Heath, 1994).The results were published in a lengthy book (Van Swaay and Warren, 2003).

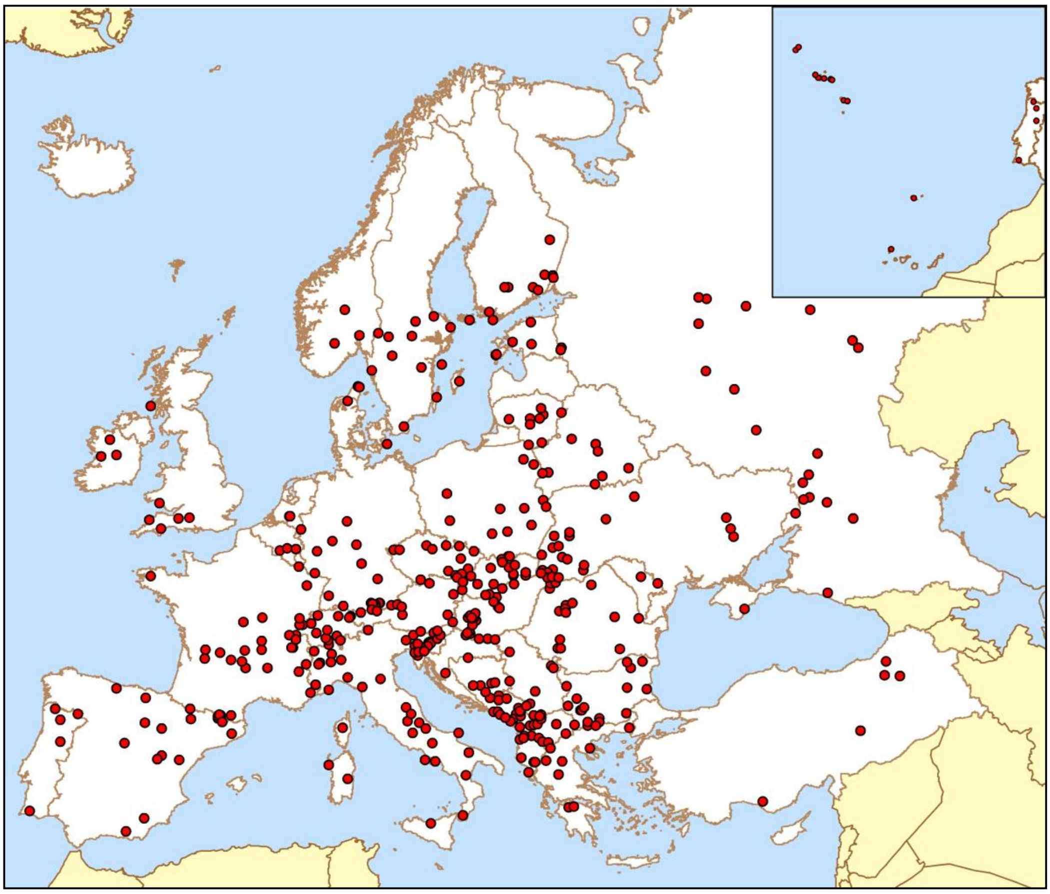

Figure 11.2. The location of the 431 Prime Butterfly Areas of Europe, identified for the 34 target species (Van Swaay & Warren, 2003).

Summary results

A total of 431 Prime Butterfly Areas were identified among 37 countries and three island archipelagos (Figure 11.2). They cover more than 21 million ha, equivalent to 1.8 % of the land area of Europe. The exact number of PBAs identified in each country depends on many different factors, such as size of the country, the number of target species present, the extent of relevant habitats remaining in the country, and the capacity to gather the data.

The most frequently occurring species within PBAs are *Maculinea arion*, *Euphydryas aurinia*, and *Parnassius apollo*, which are found in over 100 PBAs. Together with *Maculinea teleius* these three species also have the largest number of discrete breeding areas, with at least 1000 estimated populations within the PBAs. In contrast, many target species have a very restricted range and the sites selected are of the utmost importance for the conservation of rare and highly threatened species. They include several endemic species that are restricted to just one or two sites in the entire world, for example: *Gonepteryx maderensis*, *Hipparchia maderensis*, *Hipparchia azorina* ssp., *Polyommatus dama* and *P. humedasae*.

Information on trends shows that many target species are declining within PBAs, even within protected areas (Figure 11.3 and Table 11.4). This is extremely alarming and indicates that breeding habitats are deteriorating rapidly in most PBAs and that conservation measures are needed urgently. Very few species have undergone a recent increase in PBAs, the maximum being increases of Euphydryas aurinia at five sites. However trends of target species are not known for many PBAs, indicating the general need for increased monitoring of populations.

The habitat types present in PBAs reflect those of the target species present and mainly comprise woodland, alpine/sub-alpine grassland, dry grassland, and humid grassland. A great variety and intensity of land-uses are recorded within the PBAs, reflecting the long history of human settlement and management of most habitats across Europe. The conservation of habitats and butterflies therefore frequently depends on the continuation of traditional land-use practices, and suitable policies and programmes that can support them, or where necessary replace them. The main types of land-use recorded within PBAs are agriculture (62% of PBAs), forestry (60%), nature conservation (60%) and tourism and recreation (50%).

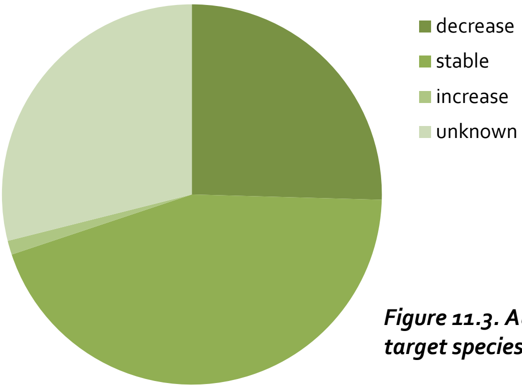

Figure 11.3. Abundance trends of target species (n=34) in PBAs.

The threats facing PBAs are diverse, ranging from adverse management activities, land-use, urban or industrial developments, and impacts of land-uses from neighbouring areas (e.g. pollution, drainage – see Figure 11.4). The main types of threats affecting PBAs are intensification of agriculture (43% of the PBAs), afforestation of former open land (40%), isolation and habitat fragmentation (35%), abandonment of traditional land use (33%, mainly in Eastern and Southern Europe). Other important threats include: adverse management, the negative effects of tourism and recreation (especially within Alpine and Mediterranean habitats), the felling of woodland, land drainage, urbanization and burning of vegetation. Collecting is not considered to be an important threat to the target species within the PBAs.
A total of 192 PBAs in Europe (44% of the total) have at least some protection under national law (Figure 11.5). In the countries of the European Union 53% of the PBAs were also classified as Natura 2000 sites. Although this large overlap of PBAs with sites protected under the Natura 2000 programme is positive, it is extremely worrying that over half of PBAs still have no international protection in spite of having major populations of butterflies for which Europe has a high responsibility.

Table 11.2: Number of reported Prime Butterfly Areas in Europe, showing trend per target species as reported by the national experts.

Species	Decrease	Stable	Increase	Unknown
Maculinea arion	42	79	2	50
Euphydryas aurinia	37	71	5	60
Maculinea teleius	29	37	1	19
Parnassius apollo	26	74	1	23
Lopinga achine	25	38		20
Maculinea nausithous	22	24	1	22
Euphydryas maturna	20	43	1	17
Coenonympha hero	16	13		13
Maculinea rebeli	10	21	1	24
Coenonympha oedippus	9	10		7
Lycaena ottomana	5	11		1
Pyrgus cirsii	3	2		12
Plebeius trappi	2	2		
Erebia sudetica	1	3		3
Pieris wollastoni	1			
Polyommatus dama	1			
Melanargia arge		6		
Erebia Christi		3		
Papilio hospiton		3		
Erebia epistygne		2		4
Erebia calcaria		2		1
Argynnis elisa		2		
Plebeius hespericus		1		4
Hipparchia miguelensis		1		1
Hipparchia maderensis		1		
Polyommatus galloi		1		
Polyommatus golgus		1		
Polyommatus humedasae		1		
Pseudochazara euxina		1		
Zerynthia caucasica		1		
Hipparchia azorina				5
Hipparchia occidentalis				2
Gonepteryx maderensis				1
Pieris cheiranthi				1

Discussion

This report documents the most important butterfly sites across Europe and we urge national conservation agencies to use the list to target protection measures within their own country and to tackle the many problems that have been identified on individual PBAs. The following specific actions are recommended:

1. Produce detailed descriptions of the PBAs within each country and designate all PBAs as protected areas under national law (NB 56 % of PBAs are not protected).
2. Protect PBAs under relevant international law such as Natura 2000 designation; and outside the EU, designation as part of the Emerald Network. (NB 47% of PBAs in the EU are not protected under international laws).
3. Provide adequate protection of PBAs in accession countries and consider PBAs identified in this review as Natura 2000 equivalent sites (eg Czech Republic, Estonia, Hungary, Poland, Slovenia and Cyprus).

4. Ensure sound habitat management within PBAs and sympathetic management in surrounding areas (e.g continuation of traditional agriculture and forestry practices and support through EC Agri-environment Regulation (EC Reg. 2078/92).
5. Take measures to conserve the wider environment and whole landscapes within and surrounding PBAs in order to sustain viable metapopulations.
6. Monitor populations of target species and conduct research to identify appropriate habitat management techniques.
7. Revise pan-European legislation urgently to take account of the new information provided in the Red Data Book of European butterflies (eg Bern Convention and the EU Habitats and Species Directive).
8. Conduct a more comprehensive review of Important Butterfly Areas in Europe as soon as possible (NB the current study has shown that this is feasible and that there is a great willingness to support such an initiative by key entomologists across Europe)
9. Keep the list of Prime Butterfly Areas up-to-date (eg via the internet)

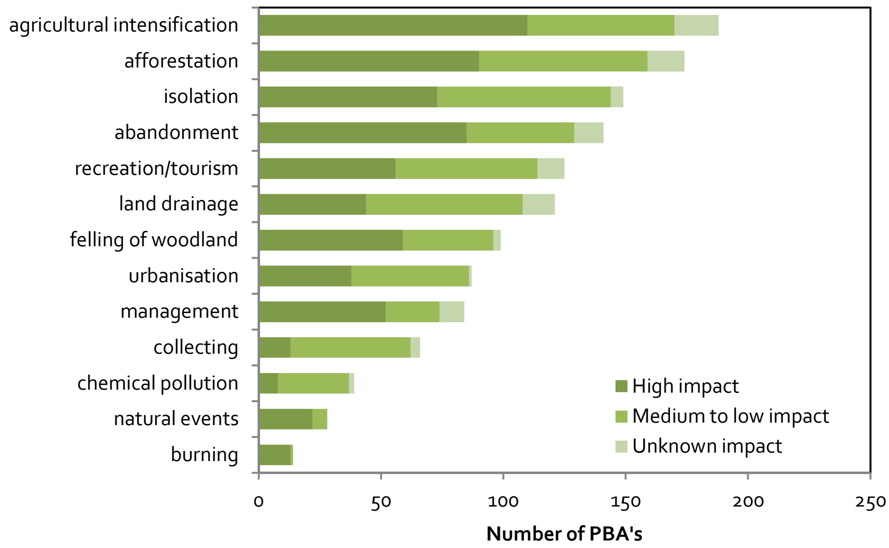

Figure 11.4. Main threats to target species within Prime Butterfly Areas.

Conclusions

The Prime Butterfly Area and Red Data Book projects demonstrate the effective collaboration of country experts to achieve shared conservation objectives over a short time scale. They have brought together unique datasets that help plan conservation at a pan European level. These datasets are already being used to initiate and guide action within many countries. However, new information on butterflies is coming available constantly and our knowledge of the status and threats to European butterflies will undoubtedly improve in coming years. We must therefore recognise that no review is ever perfect but represents a snapshot of the best data available at the time. The results also provide a good platform to build improved information systems and better conservation strategies in the future. We hope that the two projects provide useful models of what can be achieved at a European scale as similar information is needed urgently on other invertebrate taxa in order to stem their widespread decline.

The urgent need to take concerted action to conserve butterflies and moths across Europe has led us to found a new umbrella organisation in November 2004: Butterfly Conservation Europe. This aims to halt and eventually reverse the decline of Lepidoptera in Europe and promote all activities that may help to conserve this group of insects. The new organisation will co-ordinate existing work and stimulate further action both at a European policy level and at a country level. We hope to build on the successes of the two previous projects and support a growing network of organisations who are tackling Lepidoptera conservation within each country. Further details of its work can be found on www.bc-europe.eu.

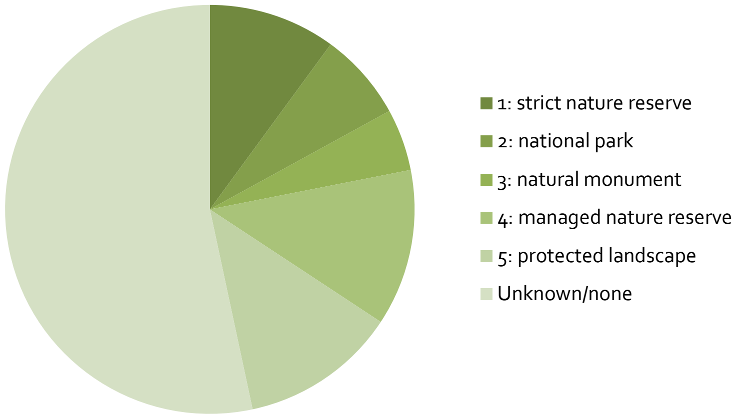

Figure 11.5. The main national protection status of Prime Butterfly Areas (n=431) in Europe. Note that PBAs can have more than one protection status.

Acknowledgements

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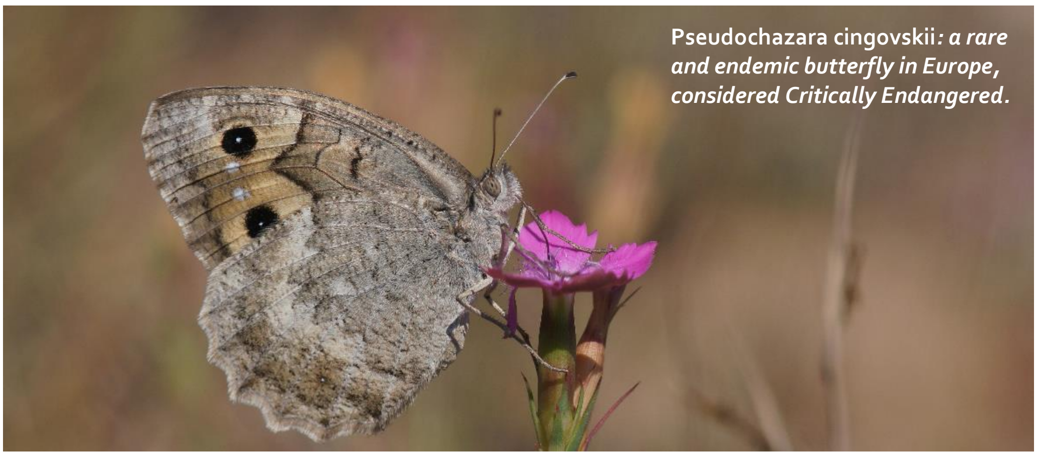
12. Applying IUCN criteria to invertebrates: How red is the Red List of European butterflies?

Slightly modified from: Van Swaay, C.A.M., Maes, D., Collins, S., Munguira, M.L., Šašic, M., Settele, J., Verovnik, R., Warren, M.S., Wiemers, M., Wynhoff, I. & Cuttelod, A. (2011)
Biological Conservation ***144 (1)**, 470-478*

Abstract

The International Union for the Conservation of Nature (IUCN) is the leading authority on assessing species' extinction risks worldwide and introduced the use of quantitative criteria for the compilation of Red Lists of threatened species. Recently, we assessed the threat status of the 483 European butterfly species, using semi-quantitative data on changes in distribution and in population sizes provided by national butterfly experts. We corrected distribution trends for the observation that coarse-scale grid cells underestimate actual population trends by 35%. If population data were unavailable, we used the distribution trend to calculate a population trend. To account for uncertainty, we included a 5% error margin on the distribution and population trends provided. The new Red List of European butterflies determined one species as Regionally Extinct, 37 species as threatened (Critically Endangered, Endangered or Vulnerable) and a further 44 as Near Threatened. The use of semi-quantitative data on distribution and population trends permitted us to use IUCN criteria to compile a scientifically underpinned Red List of butterflies in Europe. However, a comparison of detailed monitoring data for some grassland species showed that coarse-scale grid cell data and population trends strongly underestimate extinction risks, and the list should be taken as a conservative estimate of threat. Finally, combining the new Red List status with the data provided by the national butterfly experts, allowed us to determine simple criteria to delineate conservation priorities for butterflies in Europe, so called SPecies of European conservation Concern (SPEC's). Using European butterflies, our approach illustrated how Red Listing can be performed when data are incomplete for some IUCN criteria or vary strongly among countries.



Pseudochazara cingovskii*: a rare and endemic butterfly in Europe, considered Critically Endangered.*

Introduction

Since the 1950s, the International Union for Conservation of Nature (IUCN) has co-ordinated the compilation of global Red Lists, that aim to estimate the global extinction risk of each species assessed. The first Red Data Books were compiled for birds and mammals (Fitter and Fitter, 1987). Initially, categorization was based on "best professional judgment" of experts, but since the 1980s, the IUCN decided to use quantitative criteria (Mace and Lande, 1991). In 1994, the first version of the criteria and categories for compiling global Red Lists was accepted. The criteria were revised in 2001 to adapt to the needs of the various taxonomic groups (IUCN, 2001; IUCN Standards and Petitions Working Group, 2008; Mace et al., 2008). Birds were, again, the first group to which these quantitative criteria were applied (Collar et al., 1994), but since then other taxonomic groups have been evaluated as well (Baillie et al., 2004; Vié et al., 2009). The application of the IUCN criteria is, however, not always easy, especially for taxa for which quantitative data are less accurate than for mammals or birds such as bryophytes (Hallingbäck et al., 1995) or molluscs (Regnier et al., 2009). Additionally, the straightforward use of IUCN criteria on sub-global levels (Gärdenfors et al., 2001) poses some problems, especially in small regions (Maes and van Swaay, 1997). This led to the development of guidelines for the application of the IUCN Red List criteria on regional levels in 2003 (IUCN, 2003).

Butterflies are good indicators for the state of the environment and due to their short life cycle, narrow niches and relatively low mobility, they are more sensitive to land-use changes than long-lived animals such as birds and mammals (Thomas et al., 2004; Fleishman and Murphy, 2009). A further advantage of butterflies is their attractiveness to the general public, making them suitable ambassadors of biodiversity changes (Kühn et al., 2008; Schlegel and Rupf, in press). Among the invertebrates, butterflies are one of the best studied insect groups for which both ecological and relatively good quantitative distribution data are available in Europe (Kudrna, 2002; van Swaay et al., 2010). The knowledge of butterflies in Europe is fairly good compared to other parts of the world. However, differences in data quality and quantity among the different European countries still remain. In general, the countries in NW Europe usually have detailed and high quality information, but are poor in species, whereas species-rich countries in S and E Europe often have poor quality data and few people studying butterflies. This impedes the straightforward application of quantitative IUCN criteria by simply amalgamating the information of different countries.

Here, we illustrate how we applied the IUCN criteria to compile the new Red List of European butterflies. More precisely, we compared the strict quantitative use of the IUCN criteria with an approach that allows for uncertainty, correcting for underestimating the decrease in populations from coarse-scaled grid cells (Akçakaya et al., 2000). We also assessed the Red List status of 17 grassland butterflies for which detailed population data were available from butterfly monitoring schemes and compared the outcome with that of classical distribution data. Finally, based on the Red List and the additional information on distribution and population trends, we defined criteria to assess conservation priority classes for all European butterflies, so called SPecies of European conservation Concern (SPEC's).

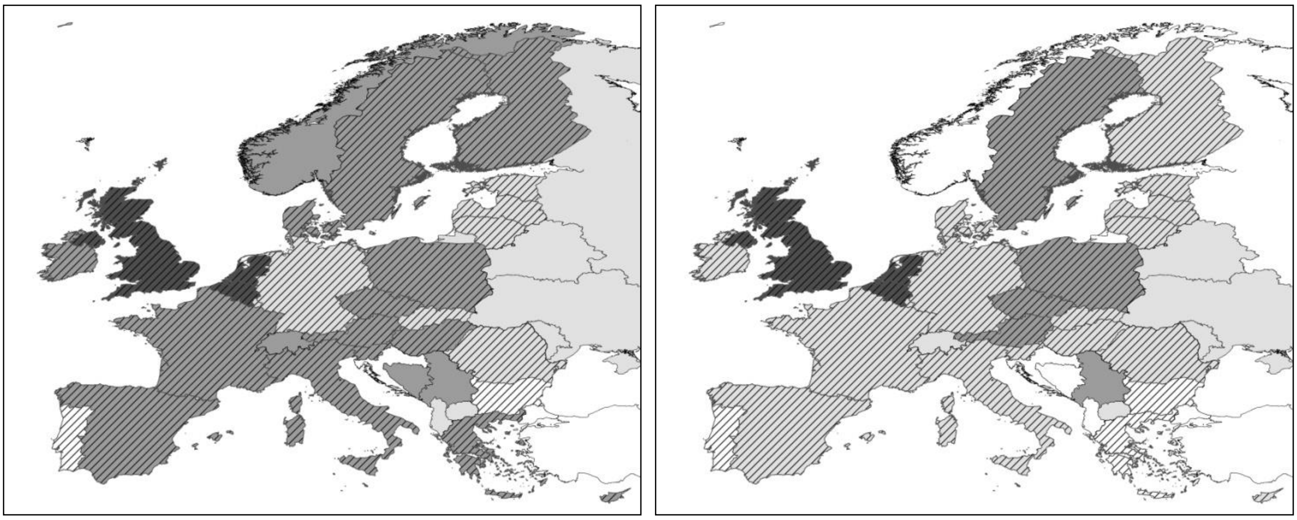


Figure 12.1: Quality of the distribution (left) and the trend (right) data as indicated by the national butterfly experts. Black = very good; dark grey = good; light grey = moderate; white = poor. The 27 countries of the European Union are hatched.

Material and methods

Geographic scope

For the purpose of this Red List, Europe ranged from Iceland to the Urals including the European parts of the Russian Federation, but excluding the Caucasus region and from Franz Josef Land to the Mediterranean, including the Canary Islands, Madeira and the Azores (Figure 12.1). Apart from the analysis for Europe as defined above, we also assessed the threat status in the 27 countries of the European Union (EU27 - Figure 12.1). The taxonomy of European butterflies was updated for the Red List (see Appendix 1) and is in concordance with the Fauna Europaea database.

IUCN criteria

The IUCN uses five criteria to assess the extinction risk of species (IUCN, 2001):

A) a past, present and/or projected population reduction measured over 10 years or 3 generations, whichever is longer; since all European butterflies have a generation time < 2 years (Tolman and Lewington, 1997) we used 10 years as a time period to estimate changes in population sizes;
B) geographic range size in combination with fragmentation, population decline or fluctuations;
C) small population size in combination with decline or fluctuations;
D) very small distribution range or restricted population size;
E) a quantitative analysis of extinction probability.

Species are assigned a threat category when satisfying any one of the criteria A-E.

Regional assessment

To determine which species to assess for the European Red List, we used the species' Range affinity, indicating the relationship between the species' European and global distribution (Kudrna, 1986; van Swaay and Warren, 1999). Species for which Europe is at the margin of their distribution range or species having only temporary populations in Europe were not assessed (Range affinity 1: 48 species – Table 12.1). As the IUCN Red List criteria are designed to be used at a global level, an adjustment is necessary to define the European Red List category for species also occurring outside Europe (IUCN, 2003). The preliminary Red List category of species for which the global distribution area is equally situated within and outside Europe (Range affinity 2) or mainly situated in Europe (Range affinity 3), can be downgraded by one category if the European populations could be rescued by populations outside Europe. This criterion is only valid for migrant or vagrant butterfly species such as Painted Lady (*Vanessa cardui*) or African Migrant (*Catopsilia florella*). Since all migrant species were categorized as Least Concern, there was no need for downgrading any of the species in the European Red List. For European endemics (Range affinity 4), the European Red List assessment coincides with the global Red List status and no adjustments of the Red List category were needed.

Data sources for the European Red List

For the compilation of the Red List of European butterflies, we used four data sources.

First, national butterfly experts provided us with the distribution area of each species in their country during the period 1999-2008 either as a percentage of the total number of investigated squares (i.e., 1 x 1 km², 5 x 5 km² or 10 x 10 km² depending on the mapping resolution used in each country) or as semi-quantitative distribution classes (<1%, 1-5%, 5-15%, >15%). For species with an Area of Occupancy (AOO) < 50 000 km² in the previous Red Data Book of European butterflies (van Swaay and Warren, 1999), additional information on the present AOO (i.e., the number of 2 x 2 km² grid cells), the number of sites, the number of individuals and the degree of fragmentation of the populations was asked. The experts also provided a distribution trend (changes in the number of grid cells) and, if available, a population trend (changes in the number of individuals) for each species for the period 1999-2008, either as an exact figure or as semi-quantitative trend classes (unknown, extinct, decrease 75-100%, decrease 50-75%, decrease 25-50%, decrease 15-25%, stable, increase 125-200%, increase >200%). Finally, the experts were asked to give an estimate of the quality of their data (Very good, Good, Moderate, Poor – Figure 12.1).

The second data source was again the previous Red Data Book of European Butterflies (van Swaay and Warren, 1999) for countries where no updated information was available. Since trends in the previous Red Data Book were based on a 25-year period, they were recalculated for a 10-year period assuming a constant exponential rate of change (i.e. constant proportional rate).

The third data source was the Climatic Risk Atlas of European Butterflies by Settele et al. (2008) in which the projected changes in the distribution of climatic niches over the period 1980-2080 were assessed. For the purpose of this Red List, we used the projected changes under the most severe (the IPCC A1FI) climate change scenario with a mean expected temperature increase of 4.1°C by 2080, assuming full dispersal. There are indications, however, that this scenario is still underestimating the negative effects of climate change on species distributions (Rahmstorf et al., 2007). Assuming unlimited dispersal is certainly over-optimistic for most butterfly species. The use of these projections of the Climatic Risk Atlas can, therefore, be seen as a conservative approach that tends to underestimate negative effects. Assuming a constant exponential rate of change (i.e. constant proportional rate) between the present-day period (1970-2000) and the period 2051-2080, we converted this projected trend into a 10-year trend to coincide with the IUCN criteria. In consultation with the IUCN, the species that were predicted to decline by 98% over 80 years (1970-2000 to 2050-2080, equivalent to <30% decline over 10 years) were classified as Near Threatened.

The final data source was the information obtained during two Red List workshops. A first workshop with 50 national butterfly experts (Laufen, Germany - January 2009) reviewed the preliminary assessments in biogeographically based subgroups. New data were added to the species summaries and maps and provisional Red List categories were defined for each species. During a second workshop (Ankara, Turkey - August 2009), we performed a sensitivity analysis on the distribution trends by adding uncertainty levels when using distribution trends to estimate the extinction risk of butterflies (cf. Akçakaya et al., 2000). Following this meeting, all butterfly assessments were reviewed and adjusted, where necessary, in consultation with the IUCN Red List Unit. The final IUCN Red List classifications can, therefore, be regarded as a product of scientific consensus through the application of semi-quantitative criteria to determine the extinction risk of all European butterflies, supported by literature and expert data sources.

Estimating the geographic range of a species in Europe

For all European butterflies, we produced distribution range maps based on European distribution data (Kudrna, 2002), national and regional atlases and European field guides (Tolman and Lewington, 1997; Lafranchis, 2004). From these maps, we calculated the Extent of Occurrence (EOO) for all European butterflies. To obtain the Area of Occupancy (AOO), we subsequently calculated the geographic range of all European species either by weighting the percentual distribution provided by the national experts (based on grid square data) or by using the geometric means of the semi-quantitative distribution classes (<1%, 1-5%, 5-15% and >15% become 0.1%, 2.24%, 8.66% and 38.73% respectively). The geographic range was calculated both for Europe as a whole and for the 27 countries of the European Union (EU27).

Estimating the European distribution trend

As mentioned above, we performed a sensitivity analysis on the distribution trend by allowing for a certain level of uncertainty on the national distribution trend provided by the national butterfly experts. In consultation with the IUCN, we decided to apply a conservative uncertainty level of 5% for all species (Akçakaya et al., 2000). If no up-to-date information on distribution changes was available, we converted the 25-year trend in the Red Data Book of European Butterflies (van Swaay and Warren, 1999) into a 10-year trend assuming a constant exponential rate of change (i.e. constant proportional rate) for the period 1999-2008 to coincide with the IUCN criteria. If only trend classes were provided in the previous Red Data Book (van Swaay and Warren, 1999), we transformed these classes in the 25-year period classes into a 10-year period trend: a decrease of 75-100% over a 25-year period was transformed into a decrease of 40-100% in the 10-year period, a decrease of 50-75% into 23-40%, a decrease of 25-50% into 10-23%, a decrease of 15-25% into 6-10%, an increase of 125-200% into 109-130% and an increase of >200% into >130%. The European distribution trend was then calculated as the mean trend of all countries weighted by their area assuming that the area of occupancy (AOO) within a country is proportional to the area of its country (van Swaay and Warren, 1999), except for the Ukraine where only the area for which data were available was used (Transcarpathia). The distribution trend was calculated both for Europe as a whole and for the EU27.

Estimating the European population trend

The procedure to establish the European population trend was similar to the distribution trend, using either the newly provided national population trends or semi-quantitative classes. Since the previous Red Data Book did not contain population trends, we used the distribution trends to derive a population trend for those countries where no such information was provided. In general, the use of 10 x 10 km² grid cells (used in the previous Red Data Book) underestimates the decrease in population trends by about 35% compared to 2 x 2 km² grid cells (Thomas and Abery, 1995; Cowley et al., 1999). Population trends are strongly correlated with trends in changes in distribution based on 2 x 2 km² grid cells (Warren et al. 2001). This resolution was also suggested by the IUCN as units for population trends (IUCN, 2001). The figure of 35% was, therefore, used to correct for the reported population trends based on 10 x 10 km² grid cells (Akçakaya et al., 2000). A decline of 10% on a 10 x 10 km² grid square basis in the previous Red Data Book, for example, was transformed into a population decline of 13.5% for the current trend calculations. The European population trend was calculated in a similar way as the European distribution trend, both for Europe as a whole and for the EU27.

Monitoring European grassland butterflies

Butterfly monitoring schemes are organised in different European countries or regions (van Swaay and van Strien, 2008) and aim to assess regional and national trends per species by means of standardized transect walks (Pollard and Yates, 1993). For 17 grassland species, monitoring data were available from 12 countries or regions: Catalunya (NE Spain), Estonia, Finland, Flanders (N Belgium), France, Germany, Jersey (Channel Islands), the Netherlands, Portugal, Argovia (N Switzerland), Transcarpathia (W Ukraine) and the United Kingdom. These data have already been used to produce the European Grassland indicator (van Swaay and van Strien, 2008). To illustrate the effect of using detailed population data (Van Dyck et al., 2009), we also calculated the IUCN Red List category for this limited set of species and compared the outcome with the classical IUCN approach described above.

SPecies of European conservation Concern

Apart from compiling a Red List for European butterflies, we also used the data to determine so called SPecies of European conservation Concern (SPEC's - van Swaay and Warren, 1999; Keller and Bollmann, 2004) based on the Red List status, endemicity and the strength of the decline (Possingham et al., 2002; Fitzpatrick et al., 2007). SPEC's can be considered as useful tools in policy making (Rodrigues et al., 2006), for example for the Bern Convention (Europe) and/or for the Habitats Directive (in the EU27). Both forms of legislation aim to protect sites for the most threatened species and/or to protect species legally.

Table 12.1: The number of butterfly species (percentages in brackets) per IUCN Red List category in Europe as a whole and for the 27 countries of the European Union (EU27). For comparison, we also give the number of species in the previous Red List (RL1999 - van Swaay and Warren, 1999). For Europe as a whole, results with and without uncertainty classes are given*, while for the EU27 only the result with uncertainty is shown. Percentages do not take into account the species that were Not Assessed or Not Evaluated. The higher number of species assesses in the 1999 Red List is due to the fact that Asian Turkey was included in the analysis.

IUCN Category	Europe Without uncertainty	With uncertainty	EU27	RL1999
Regionally Extinct	1 (0.2)	1 (0.2)	2 (0.5)	1 (0.2)
Critically Endangered	3 (0.7)	3 (0.7)	2 (0.5)	7 (1.2)
Endangered	8 (1.8)	12 (2.8)	9 (2.1)	18 (3.1)
Vulnerable	11 (2.5)	22 (5.1)	19 (4.5)	45 (7.8)
Near Threatened	25 (5.7)	44 (10.1)	47 (11.2)	42 (7.3)
Least Concern	383 (88.0)	349 (80.2)	337 (80.2)	463 (80.4)
Data Deficient	4 (0.9)	4 (0.9)	4 (1.0)	-
Not Applicable or Not Evaluated	48	48	63	-
Species assessed	435	435	420	576

**The consulted experts could either report the trend as a an exact figure (e.g. 'decline of 20%') or as semi-quantitative trend classes (e.g. 'decrease of 15-25%'). After introducing a 5% uncertainty in the exact figures and using the class-borders, the minimum and maximum trend could be calculated lead. Following the precautionary principle the largest decline was used to classify the species for the columns 'with uncertainty'.*

Table 12.2: IUCN Red List category for 17 grassland butterflies using trends based on distribution data and on monitoring data.

Species	Distribution data	Monitoring data
Anthocharis cardamines	Least Concern	Least Concern
Coenonympha pamphilus	Least Concern	Near Threatened
Cupido minimus	Least Concern	Least Concern
Cyaniris semiargus	Least Concern	Vulnerable
Erynnis tages	Least Concern	Near Threatened
Euphydryas aurinia	Least Concern	Least Concern
Lasiommata megera	Least Concern	Near Threatened
Lycaena phlaeas	Least Concern	Least Concern
Maniola jurtina	Least Concern	Least Concern
Ochlodes sylvanus	Least Concern	Least Concern
Phengaris arion	Endangered	Critically Endangered
Phengaris nausithous	Near Threatened	Least Concern
Polyommatus bellargus	Least Concern	Least Concern
Polyommatus coridon	Least Concern	Least Concern
Polyommatus icarus	Least Concern	Least Concern
Spialia sertorius	Least Concern	Least Concern
Thymelicus acteon	Near Threatened	Vulnerable

Results

Data quality, as indicated by the national butterfly experts differed considerably over Europe (Figure 12.1). Distribution data were, on average, estimated as relatively good (2.46 on a scale from 1 very good to 4 poor), while, on average, trend data were qualified as moderate (average 2.89).
The strict quantitative use of the IUCN criteria would have classified 23 butterflies as Regionally Extinct or threatened (Critically Endangered, Endangered or Vulnerable - Table 12.1). Allowing for uncertainty in the population trends resulted in the classification of 38 species as Regionally Extinct or threatened (Table 12.1). Using the detailed population trends from the monitoring transects would have placed six of the 17 grassland species in a higher and one in a lower threat category (Table 12.2). Although IUCN permits the use of the best information available to estimate extinction risks, we preferred to evaluate all species using distribution or population trends, because monitoring data are biased towards NW Europe. The final IUCN Red List category of all 483 European species together with the IUCN criteria used to classify the species can be found in Van Swaay et al. (2010).
In the new Red List of European butterflies, Anatolian False Argus (*Aricia hyacinthus*) is the only species classified as Regionally Extinct. It was known only from SW Romania in the beginning of the 20th century (Székely, 2008) and the nearest present-day populations are in W Turkey (Hesselbarth et al., 1995). 37 species (9%) were classified as threatened and a further 44 species (10%) as Near Threatened (Table 12.1). The three Critically Endangered species in Europe are: Maderian Large White (*Pieris wollastoni* - restricted to the island of Madeira, but probably extinct because it has not been observed since the 1980s), Siberian Brown (*Coenonympha phryne* - only present in the Ukraine on two small virgin steppes in NE Crimea and in Russia reported as extremely rare) and Macedonian Grayling (*Pseudochazara cingovskii* - known from only one location of less than 1.5 km² in Macedonia).

Table 12.3a: Conservation priority classes of the SPecies of European conservation Concern (SPEC's) in the whole of Europe. The numbers shown in brackets is the number of species in each conservation priority class. (II) = the species is already listed in the Bern Convention Annex II.

Regionally Extinct in Europe Critically Endangered in Europe Endangered in Europe Vulnerable in Europe SPEC1	Near Threatened in Europe SPEC2	Decline in >=35% of the Europen countries Decline >10% SPEC3	 Decline <10% SPEC4
European endemics (22)	**European endemics (11)**	**European endemics (3)**	**European endemics (3)**
Coenonympha orientalis	*Colias phicomone*	*Erebia manto*	*Melitaea parthenoides*
Erebia christi (II)	*Erebia claudina*	*Hipparchia maderensis*	*Plebejus hespericus*
Erebia sudetica (II)	*Erebia epistygne*	*Melitaea asteria*	*Pseudophilotes baton*
Euchloe bazae	*Erebia flavofasciata*		
Gonepteryx cleobule	*Hipparchia fagi*	**Non-endemics (19)**	**Non-endemics (6)**
Gonepteryx maderensis	*Hipparchia leighebi*	*Argynnis niobe*	*Boloria frigga*
Hipparchia bacchus	*Hipparchia sbordonii*	*Boloria freija*	*Colias palaeno*
Hipparchia tilosi	*Plebejus trappi*	*Colias tyche*	*Euphydryas aurinia* (II)
Pararge xiphia	*Polyommatus nephohiptamenos*	*Erebia embla*	*Oeneis bore*
Pieris cheiranthi	*Polyommatus nivescens*	*Glaucopsyche alexis*	*Oeneis jutta*
Pieris wollastoni	*Pseudophilotes panoptes*	*Hamearis lucina*	*Plebejus sephirus*
Plebejus zullichi		*Hesperia comma* (II)	
Polyommatus galloi	**Non-endemics (33)**	*Hyponephele lycaon*	
Polyommatus golgus (II)	*Archon apollinus*	*Lycaena hippothoe*	
Polyommatus humedasae (II)	*Aricia anteros*	*Lycaena thersamon*	
Polyommatus orphicus	*Boloria chariclea*	*Melanargia occitanica*	
Polyommatus violetae	*Boloria titania*	*Neolycaena rhymnus*	
Pseudochazara amymone	*Carcharodus flocciferus*	*Nymphalis vaualbum*	
Pseudochazara cingovskii	*Carcharodus lavatherae*	*Papilio alexanor*	
Pseudochazara euxina	*Chazara briseis*	*Phengaris alcon*	
Pseudochazara orestes	*Colias hecla*	*Pseudophilotes bavius*	
Pyrgus cirsii	*Cupido decoloratus*	*Pyronia tithonus*	
	Euphydryas desfontainii	*Satyrium ilicis*	
Non-endemics (16)	*Euphydryas iduna*	*Tomares callimachus*	
Aricia hyacinthus	*Hipparchia hermione*		
Boloria improba	*Hipparchia statilinus*		
Boloria polaris	*Iolana iolas*		
Coenonympha hero (II)	*Leptidea morsei*		
Coenonympha oedippus (II)	*Maniola halicarnassus*		
Coenonympha phryne	*Melitaea aurelia*		
Coenonympha tullia	*Melitaea britomartis*		
Colias chrysotheme	*Muschampia cribrellum*		
Colias myrmidone	*Oeneis norna*		
Euphydryas maturna (II)	*Parnassius apollo* (II)		
Lopinga achine (II)	*Parnassius mnemosyne* (II)		
Lycaena helle	*Parnassius phoebus*		
Phengaris arion (II)	*Phengaris nausithous* (II)		
Phengaris teleius (II)	*Plebejus dardanus*		
Tomares nogelii	*Plebejus pylaon*		
Turanana taygetica	*Polyommatus damon*		
	Polyommatus dorylas		
	Polyommatus eros		
	Pseudophilotes vicrama		
	Thymelicus acteon		
	Zegris eupheme		
	Zerynthia cerisy		

Table 12.3b: Conservation priority classes of the SPecies of European conservation Concern (SPEC's) in the 27 countries of the European Union (EU27). The figure shown in brackets is the number of species in each conservation priority class. (II), (IV), (II/IV) = the species is already listed in the Habitat Directive Annex II or IV or both.

Regionally Extinct in EU27 Critically End. in EU27 Endangered in EU27 Vulnerable in EU27 SPEC1	***Near Threatened in EU27 SPEC2***	***Decline in >=35% of the EU27 countries Decline >10% SPEC3***	***Decline <10% SPEC4***
European endemics (19)	**European endemics (10)**	**European endemics (6)**	**European endemics (2)**
Erebia christi (II/IV)	*Colias phicomone*	*Erebia manto*	*Erebia nivalis*
Erebia sudetica (II/IV)	*Erebia claudina*	*Erebia melas*	*Plebejus hespericus*
Euchloe bazae	*Erebia epistygne*	*Hipparchia maderensis*	
Gonepteryx cleobule	*Erebia flavofasciata*	*Hipparchia semele*	**Non-endemics (17)**
Gonepteryx maderensis	*Hipparchia fagi*	*Oeneis glacialis*	*Argynnis aglaja*
Hipparchia bacchus	*Hipparchia leighebi*	*Pyrgus warrenensis*	*Boloria euphrosyne*
Hipparchia tilosi	*Hipparchia sbordonii*		*Boloria frigga*
Pararge xiphia	*Polyommatus nephohiptamenos*	**Non-endemics (16)**	*Boloria selene*
Pieris cheiranthi	*Polyommatus nivescens*	*Boloria freija*	*Colias palaeno*
Pieris wollastoni	*Pseudophilotes panoptes*	*Boloria thore*	*Colias tyche*
Plebejus zullichi	**Non-endemics (37)**	*Coenonympha oedippus*	*Cyaniris semiargus*
Polyommatus galloi	*Argynnis laodice*	*Erebia aethiops*	*Erebia ligea*
Polyommatus golgus (IV)	*Argynnis niobe*	*Erebia embla*	*Erebia medusa*
Polyommatus humedasae	*Boloria chariclea*	*Euphydryas maturna* (II/IV)	*Erynnis tages*
Polyommatus orphicus	*Carcharodus lavatherae*	*Hesperia comma* (II)	*Euphydryas aurinia* (II)
Polyommatus violetae	*Chazara briseis*	*Lycaena helle* (II/IV)	*Hamearis lucina*
Pseudochazara amymone	*Coenonympha tullia*	*Melanargia occitanica*	*Lycaena virgaureae*
Pseudochazara orestes	*Colias hecla*	*Melitaea aurelia*	*Oeneis bore*
Pyrgus cirsii	*Euphydryas desfontainii*	*Melitaea cinxia*	*Oeneis jutta*
	Euphydryas iduna	*Parnassius mnemosyne* (II/IV)	*Pyrgus centaureae*
Non-endemics (13)	*Hipparchia hermione*	*Polyommatus admetus*	*Pyrgus malvae*
Aricia hyacinthus	*Hipparchia statilinus*	*Polyommatus bellargus*	
Boloria improba (II)	*Iolana iolas*	*Pyrgus armoricanus*	
Boloria polaris	*Limenitis populi*	*Satyrium ilicis*	
Coenonympha hero (II/IV)	*Lycaena alciphron*		
Colias chrysotheme	*Lycaena hippothoe*		
Colias myrmidone (II/IV)	*Maniola halicarnassus*		
Leptidea morsei (II/IV)	*Melitaea britomartis*		
Lopinga achine (IV)	*Melitaea diamina*		
Nymphalis vaualbum (II/IV)	*Melitaea trivia*		
Phengaris arion (II/IV)	*Muschampia cribrellum*		
Phengaris teleius (II/IV)	*Nymphalis xanthomelas*		
Tomares nogelii	*Oeneis norna*		
Turanana taygetica	*Parnassius apollo* (II/IV)		
	Parnassius phoebus		
	Phengaris alcon		
	Phengaris nausithous (II/IV)		
	Plebejus dardanus		
	Polyommatus damon		
	Polyommatus dorylas		
	Polyommatus eros (II/IV)		
	Polyommatus ripartii		
	Pseudophilotes vicrama		
	Pyrgus serratulae		
	Scolitantides orion		
	Thymelicus acteon		
	Zegris eupheme		
	Zerynthia cerisy		

In the EU27, two species were classified as Regionally Extinct: Anatolian False Argus and Nogel's Hairstreak (*Tomares nogelii*), both reported only from Romania. Two species are Critically Endangered in the EU27: Maderian Large White (see above) and Danube Clouded Yellow (*Colias myrmidone*), the latter being one of the most rapidly declining species in the region (Dolek et al., 2005; Konvicka et al., 2008). Thirty species (7%) were considered threatened and a further 47 (11%) Near Threatened (Table 12.1).

The most often used criteria to classify European butterflies in their final Red List category were criterion A (declining populations – 47 times) and criterion B (restricted geographic range size, and fragmentation, decline or fluctuations – 22 times). Criterion C (small population size and decline) was only used for Spanish Greenish Black-tip (*Euchloe bazae*), an extremely local species occurring only in Spain; criterion D (very small population or very restricted distribution) was used to classify six species: El Hierro Grayling (*Hipparchia bacchus*), two locations on El Hierro in the Canary islands (Spain); La Palma Grayling (*Hipparchia tilosi*), five locations on La Palma in the Canary islands (Spain); Nevada Blue (*Polyommatus golgus*), nine locations with an area of occupancy (AOO) of 16 km² in Sierra Nevada and Sierra de la Sagra (Spain); Andalusian Anomalous Blue (*Polyommatus violetae*), two locations in the Sierras of Almijara, Tejeda, Cazorla and La Sagra (Spain), Brown's Grayling (*Pseudochazara amymone*), four locations in NW Greece and Dils' Grayling (*Pseudochazara orestes*), five locations along the border between Greece and Bulgaria.

The SPecies of European conservation Concern with the highest conservation priority (SPEC1) are the Regionally Extinct and threatened species (38 and 32 species in the whole of Europe and in the EU27 respectively – Table 12.3). A second conservation priority (SPEC2) concerns all Near Threatened species (44 and 47 species in Europe and in the EU27 respectively – Table 12.3). Additionally, species that are of Least Concern, but show a declining population trend in at least 35% of the countries, consist a third (overall decline >10% - SPEC3) and fourth (overall decline <10% - SPEC4) conservation priority (Table 12.3).

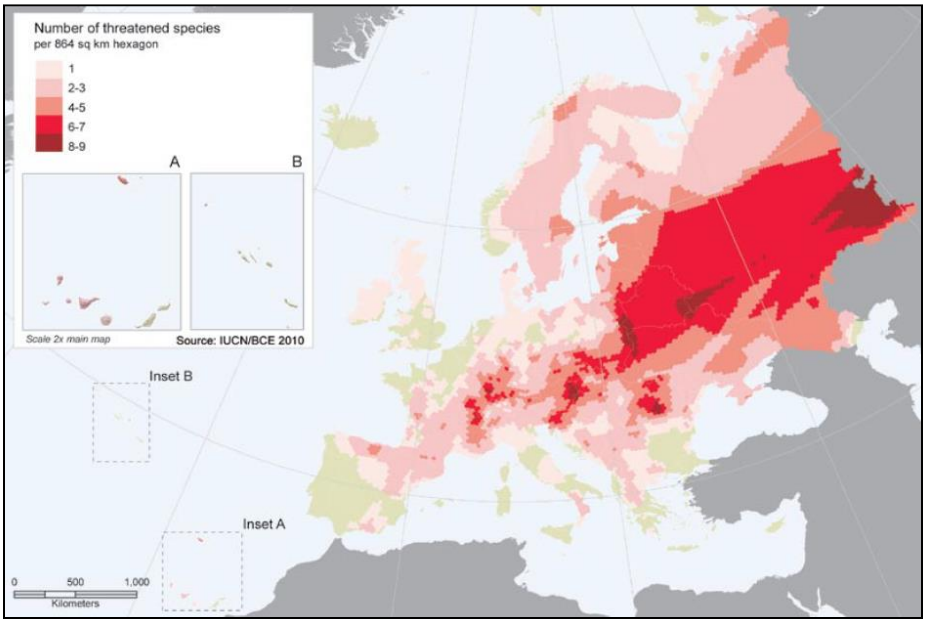


Figure 12.2: Number Regionally Extinct, Critically Endangered, Endangered or Vulnerable butterfly species per 864 km² hexagon in Europe.

Discussion

Despite regional differences in data quality, we managed to apply quantitative IUCN criteria to estimate the extinction risk of European butterflies. Allowing for uncertainty in data accuracy (Akçakaya et al., 2000) and applying the precautionary principle (Kriebel et al., 2001), we classified about 19% of all European butterflies as threatened (i.e., Critically Endangered, Endangered or Vulnerable) or Near Threatened. The use of detailed monitoring data for a limited set of grassland species, however, showed that coarse-scale grid cell data and population trends strongly underestimate extinction risks. The new European Red List of butterflies should, therefore, be seen as rather conservative (i.e., only listing clearly threatened species). However, it is also an opportunity to demonstrate the rapid loss of European butterflies and the need for urgent conservation action.

Data quality and the use of IUCN criteria for European butterflies

Data quality differed considerably among countries and also between distribution and trend data. 53% of the European countries estimated their *distribution* data as good, while only 21% estimated their *trend* data to be good. Since self-assessment of data quality is often subjective (Burgman 2005), it would – for the future – be preferable to assess the data quality of (national) surveys with a more objective measure, both in space and in time. NW European countries generally have high quality data on butterfly distribution and population trends. For example, Britain (Asher et al., 2001), the Netherlands (Bos et al., 2006) and Belgium (Maes and Van Dyck, 2001; Fichefet et al., 2008) all have recent distribution atlases and/or butterfly monitoring schemes. In contrast, many E and SE European countries, but also countries such as Germany, France or Italy, have far less detailed distribution and trend data (Figure 12.1). The use of distribution and trend classes, however, allowed us to adequately include all countries in the Red List assessment of European butterflies. Since Central and E Europe are strongholds for many of the threatened butterflies in Europe (Figure 12.2 – van Swaay et al., 2010), it is important to gather more detailed data in these countries. To improve the quality and the quantity of data necessary for compiling European but also national Red Lists, we would encourage all European countries to start collecting detailed butterfly distribution data and, where possible, to start a robust butterfly monitoring scheme (e.g. Kühn et al., 2008). The use of skilled volunteers can make these schemes relatively cheap and will rapidly increase the number of records (Schmeller et al., 2008).

The use of the Climatic Risk Atlas of European Butterflies (Settele et al. 2008) enabled us to classify nine species in the Near Threatened category: four arctic species (Arctic Fritillary *Boloria chariclea*, Northern Clouded Yellow *Colias hecla*, Lapland Fritillary *Euphydryas iduna* and Norse Grayling *Oeneis norna*) and five species that, in Europe, mainly occur in the southwest (Spring Ringlet *Erebia epistygne*, Spanish Fritillary *Euphydryas desfontainii*, Mother-of-pearl Blue *Polyommatus nivescens*, Panoptes Blue *Pseudophilotes panoptes* and Sooty Orange-tip *Zegris eupheme*). However, the climate risk atlas assessed 50 more species as having an extremely high risk of going extinct due to climate change (i.e., a loss of more than 95% in distribution area). If the climate change criterion would have been lowered to a 95% distribution loss (as in Settele et al. 2008) instead of the 98% used here, 44 additional species would have been classified as Near Threatened in the present European Red List. Climate change atlases that estimate the extinction risk of species such as the ones for birds (Huntley et al. 2007) and butterflies (Settele et al. 2008) are powerful conservation tools and can, as shown here, be used as criterion A3c (i.e., a projected population reduction over 10 years) in the Red Listing process.

The use of coarse-scale grid cells as units of species' distribution strongly overestimates the area occupied by a species (Thomas and Abery, 1995; Cowley et al., 1999), leading to an underestimation of the decline in distribution. For example, detailed measurements of the distribution of Alcon Blue (*Phengaris alcon*) in Belgium revealed that it actually occupied only 0.48 km² (Maes et al., 2004). When expressing its distribution in grid cells, however, it occurs in 22 grid cells of 1 x 1 km² (22 km²), 15 cells of 2 x 2 km² (60 km²), 13 cells of 5 x 5 km² (325 km²) and nine cells of 10 x 10 km² (900 km²). This would overestimate the AOO by a factor of 46, 125, 677 and 1875 respectively. To minimise the overestimation of a species' distribution and to produce estimates of the area of occurrence (AOO) that are valid for comparison with the thresholds in criterion B, the IUCN recommends the use of 2 x 2 km² grid cells (IUCN Standards and Petitions Working Group, 2008). Furthermore, where actual population trends from monitoring schemes were available, they showed that declines in population trends based on distribution data are even more strongly underestimated. On the other hand, a misjudgement of a single national expert might easily result in a very local species being listed as threatened without proper justification. Brown grayling (*Pseudochazara amymone*) and Dil's grayling (*Pseudochazara orestes*), for example, are restricted to Greece and/or Bulgaria, two countries without a butterfly monitoring scheme able to estimate population trends of this local endemics. Despite this, both species are classified as Vulnerable, based on the opinion of local experts.

One of the major problems in applying the IUCN criteria to butterflies is the 10-year period which is rather short to detect declines of more than 30% to classify the species at least as Vulnerable. Having one or even more generations per year and being more sensitive to environmental factors, invertebrate numbers tend to fluctuate much more than those of long-lived animals (Thomas, 1994). This makes it difficult to distinguish between anthropogenically induced declines and natural fluctuations. Moreover, the IUCN criteria are designed to estimate a species imminent extinction risk and do not take declines in an earlier period into account. Although understandable, it may lead to very counterintuitive classifications. There are several examples of species (mostly habitat specialists) that have almost disappeared from W and C Europe in the second half of the 20th century (resulting in a population decline of more than 80%), but the few remaining populations have been maintained and are either stable or show slow declines of less than 30% partly as a result of huge conservation efforts. Examples include Moorland Clouded Yellow (*Colias palaeno* - Nilsson et al., 2008) and Cranberry Fritillary (*Boloria aquilonaris* - Baguette and Schtickzelle, 2003) in peat bogs, False Ringlet (*Coenonympha oedippus* - Lhonoré and Lagarde, 1999; Čelik et al., 2009) and Violet Copper (*Lycaena helle* - Bauerfeind et al., 2009) in wetlands and Scarce Fritillary (*Euphydryas maturna* - Cizek and Konvicka, 2005) in coppiced woodlands. Because of a strong decline in the second half of the 20th century, False Ringlet, for example, was listed as one of the most threatened species in the previous Red Data Book of European butterflies (van Swaay and Warren, 1999) but is now classified as Least Concern in the EU27. Most of these species used to be much more common and widespread in Europe, and are thought to belong to the most threatened species by many butterfly experts. However, since they declined by less than 30% in the last 10 years, they were classified as non-threatened in the present Red List. The number of individuals in populations of these species may appear stable to experts visiting the sites on an irregular basis. Over several decades, however, such species have often become extinct across large regions (even in nature reserves) due to gradual but permanent habitat deterioration and natural fluctuations. Since such fluctuations are of much higher amplitude in insects than in vertebrates, they can more easily lead to extinctions. Considering a

longer time period (e.g., 25 years) to estimate extinction risks, would, therefore, be advisable for butterflies and possibly also for other invertebrates.
A straightforward comparison of the Red Data Book (van Swaay and Warren, 1999) and the present Red List (van Swaay et al. 2010) is not possible due to differences in methodology, spatial extent and timescales used between both lists. Compared to the previous Red List, a smaller number of species was now categorized as threatened (Table 12.1).However, applying the methodology of 1999 to the present data would have classified 112 species in a higher and 37 in a lower Red List category than with the latest IUCN methodology (analysis not shown but see van Swaay and Warren (1999)). Using the 1999 methodology, 19 (4%) more species would have been classified in the present Red list as Regionally Extinct or threatened (Critically Endangered, Endangered or Vulnerable) and 49 (10%) as Near Threatened. The lower number of threatened species in the present list can, therefore, at least partly be attributed to the more conservative new IUCN criteria. Other methodological differences, e.g. the fact that Asian Turkey was included in the previous Red List, also make it impossible to compare both lists (Keith and Burgman, 2004).

Conservation and policy implications
With about 9% of the European butterflies classified as threatened and an additional 10% as Near Threatened, our results show that declines in butterfly diversity have certainly not been halted. In total, we determined 113 and 120 SPEC's for Europe and for the EU27 respectively (Table 12.3). Twenty-two of these species are already on the Bern Convention or Habitat Directive Annexes (Table 12.3). For all SPEC's, we would recommend the compilation of species action plans describing the causes of decline and the management and policy actions required (e.g., Munguira and Martín, 1999). An underpinned Red List and improved legislation protecting both suitable biotopes and species are, therefore, both important to stimulate butterfly conservation in Europe. The protection of sites of the most threatened butterfly species could focus on so called Prime Butterfly Areas (PBA's), that are delineated on the basis of a target-species list of butterflies (European importance, conservation priority, etc). The list of SPEC's presented here would be a good complement to the list of species that was used to delineate the present PBA's because it involves more species (113 instead of 34), more specific biotopes (e.g., tundra, mountain areas) and more biogeographical regions (especially N European species are under-represented in the present target-species PBA list). PBA's have already been described for Europe (van Swaay and Warren, 2003), have recently been updated for Bulgaria (Abadjiev and Beshkov, 2007) and Serbia (Jakšić, 2008) and a revision is being prepared for Turkey.

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13. Synthesis

Introduction

In the preservation of biodiversity, butterfly conservation plays a major role in representing the large group of insects. Apart from one or two other groups (e.g. dragonflies and grasshoppers) there are no insect groups for which large-scale information on distribution and trends are available, as well as knowledge on their ecology and conservation.
In chapter 1 the five major pillars in butterfly conservation were described:

1. distribution
2. trend
3. causes
4. conservation
5. communication

Three of these pillars have been investigated further in this thesis. In the first part (chapters 2-4), the focus was on establishing the distribution and especially trends in the distribution of species. Trends in distribution can be significantly different from the trend in population size, which is the topic in the second part (chapters 5-8). Using this information to gather more knowledge on the conservation of butterflies makes up the last part (chapters 9-12).

Challenges in tracking changes in butterfly distribution

In their basic form, distribution maps show dots which represent observations of species. In most cases these dots are displayed in some kind of grid system and summarise the recording over periods of time. However, such maps can be difficult to interpret, as there are large differences in the periods, research intensity, scale of observations etc. (chapter 2). In the worst case, these maps even don't show the distribution of a species but something completely different, for example the distribution of recorders or of train stations which could easily be reached.
Although the number of recorders, both in the Netherlands, Europe and the world, has risen considerably, these facts still play an important role in compiling distribution maps, even of relatively well-investigated groups like butterflies: the higher demand for high quality and detailed information in landscape planning, nature conservation and management has more or less compensated for the rise in the number and the precision of butterfly records. Where, for Lempke (1936), a list of municipalities was more than sufficient, without any information on the date of the observation or on the number of records, and Geraedts (1986) and Tax (1989) were quite satisfied with squares of 5 x 5 km, even a 1 x 1 km grid is now considered to be on the coarse side (figure 13.1). Nowadays, online maps and in-the-field-recording on a smartphone with gps improve the precision to under 10 m.

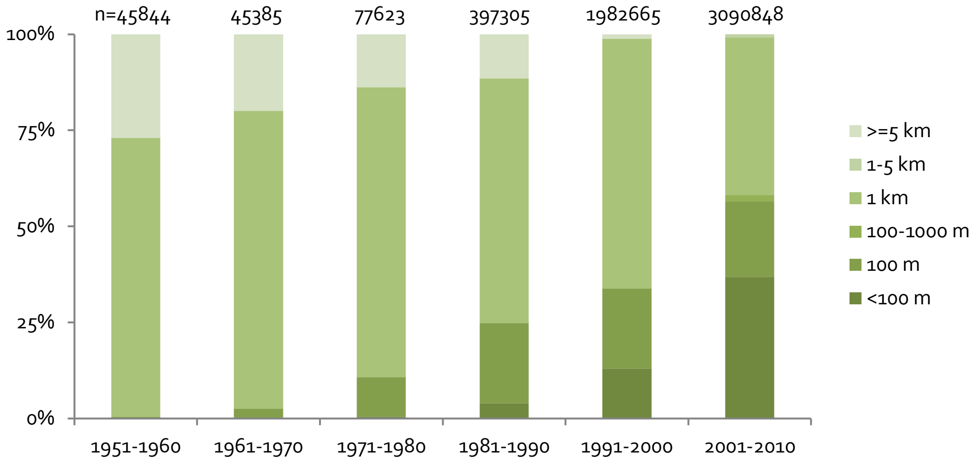


Figure 13.1: Quality of the distribution data of butterfly records in the Dutch National Database Flora and Fauna (NDFF). The totol number of butterfly observations in each period is given on the upperside of the bars.

Although large numbers of recorders potentially reduce the gaps of knowledge in the maps, it is still essential to be able to judge the value of records as well as the squares with missing values. Several methods have been developed to tackle these issues:

- Probability map: a map showing the probability that a species occurs in a square, mostly built on models using abiotic and habitat information (Van Swaay et al., 2006).
- Gap closure: filling a distribution map (a map with all positive records) to a range map using a predefined set of rules. This method is used by the European Union for completing and unifying distribution maps at a European scale for the reporting on species of the annexes of the Habitats Directive under article 17 (Evans & Arvela, 2011).
- Occupancy modelling: correct for the detection probability of species in grid cells (chapter 4).

Probability maps

The distribution of species is determined by a complex of interacting factors. If all factors are known and all relationships parameterised in a model, a complete distribution map should theoretically be possible. Such models usually end up with a probability of a species to occur at a given site given the combination of ecological parameters. To produce such probability maps, distribution records are linked to available data on abiotic parameters as well as habitat information or even the distribution of other species (Maes et al. 2009). The quality of the maps is determined by the models used, the number of parameters as well as the quality of the parameter maps and the number and quality of the detection/non-detection data.

In the Netherlands, such maps have so far been produced for butterflies on two occasions. In 2006 a report on biodiversity hotspots for butterflies in the Netherlands was produced (Van Swaay et al. 2006) with probability maps for all species on a level of 250 x 250 m for the period 2000-2005. Figure 13.2 shows the probability distribution of *Hipparchia semele* on a 250 x 250 m scale as well as summarised on a 5 x 5 km scale.

More recently Sierdsema (pers. comm.) is in the process of producing probability maps for all protected Dutch species including butterflies.

Although a useful tool, probability maps suffer from the fact that the final result heavily depends on the quality and update frequency of the underlying data-maps. As there are always some maps that are not or only infrequently updated (which means some of them can be more than ten years old), probability maps have a serious risk of being already outdated when produced or soon after.

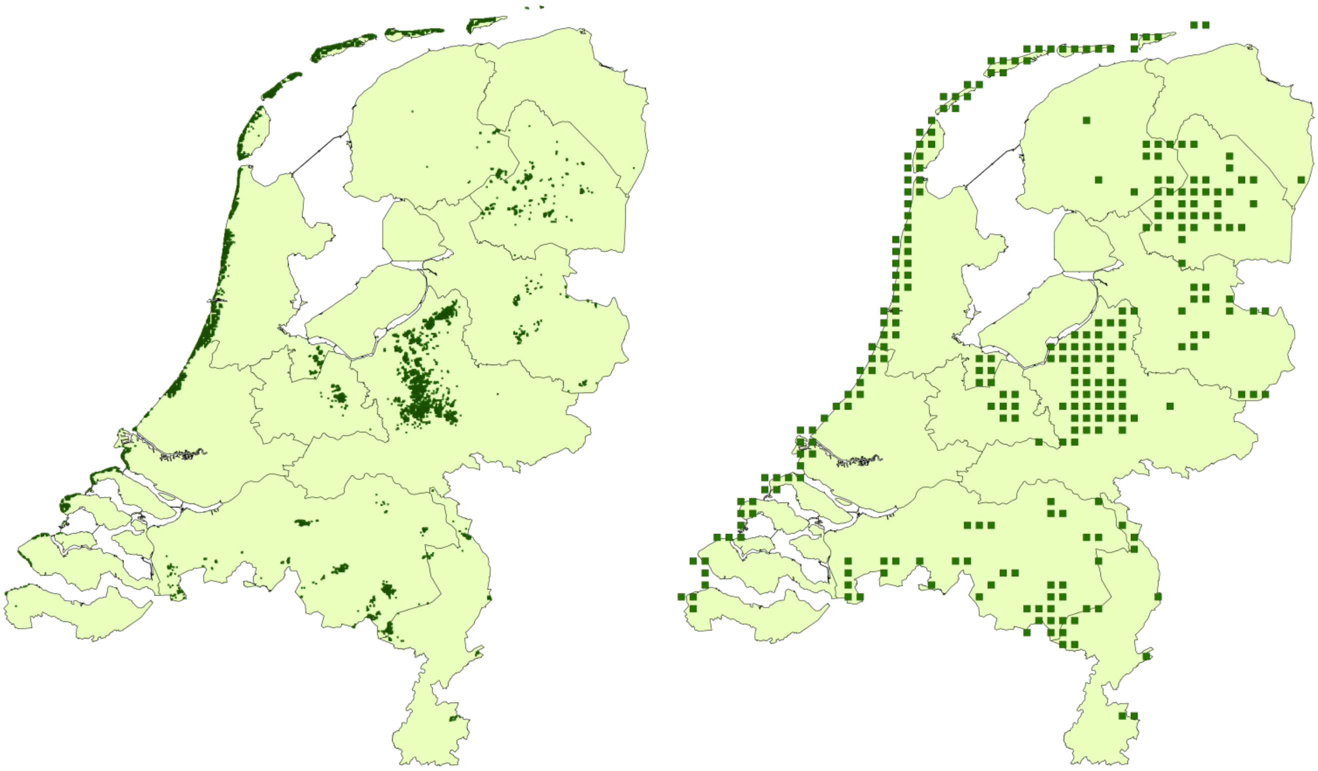
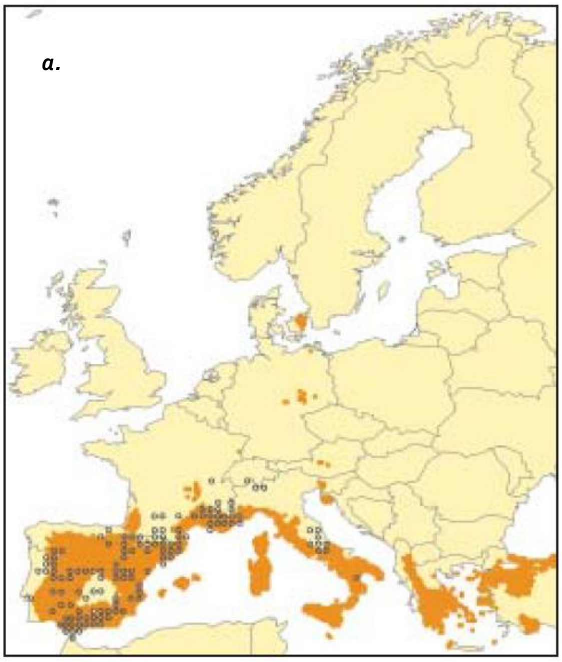


Figure 13.2: Probability maps, indicating suitable grid cells, for **Hipparchia semele** ***in the Netherlands in the period 2000-2005 on a 250x250m grid (left) and summarized to a 5x5km grid (right) (Van Swaay et al., 2006). Source: NDFF.***

At a larger scale, butterfly distributions heavily depend on climatic circumstances. As a consequence, it has proved possible to generate climatic niche models producing European Climate Envelopes per species describing the distribution of species at a European scale and based on four climatic variables (Settele et al., 2008):

- accumulated growing degree days until August, which is highly representative for general temperature gradients across Europe;
- soil water content for the upper horizon, which is a realistic measure of water availability and near surface microclimate;
- ranges in annual precipitation;
- ranges in annual temperature;

with the two last variables reflecting continentality and oceanity.
The resulting probability maps can be extended to the future by applying possible climate scenarios (see figure 13.3 for an example). Such climate envelopes and the resulting maps not only help in understanding the relationships between butterflies and the main climatic drivers, a risk analysis also makes it possible to anticipate on future changes and take measures to counteract negative implications.



Male (right) and female of Anthocharis euphenoides.

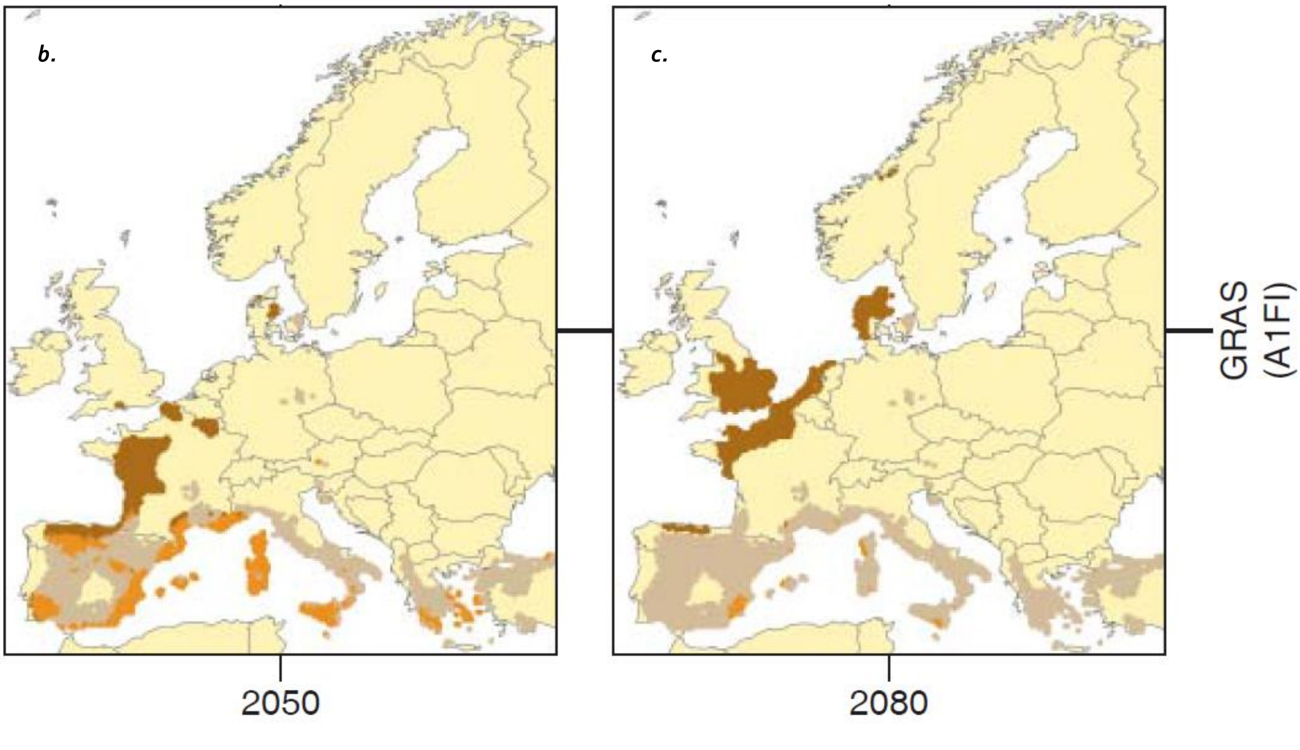


Figure 13.3: *Observed and modelled actual distribution as well as potential future distributions in 2050 and 2080 under the GRAS (GRowth Applied Strategy) scenario (approximating the IPCC A1FI climate change scenario with mean expected increase in temperature of 4.1°C) (Settele et al., 2008):*

a. *Observed species distribution (50 × 50 km² UTM grid; black circles) and modelled actual distribution of climatic niche (orange areas) of* Anthocharis euphenoides *in Europe.*
b. *Potential distribution under the GRAS scenario in 2050 (orange= remains stable; grey= gets lost, and dark brown= gained.*
c. *Potential distribution in 2080.*

Gap closure

Gap closure is a method which is used to produce range maps for species and habitats, listed on the annexes of the Habitats Directive of the European Union, for the reporting following article 17 of that directive. Gap closure is described as: 'using a predefined set of rules specifying when two distribution points/grids will be joined together to form a single range polygon and where an actual gap in the range will be left' (Evans and Arvela, 2011). This is a useful tool on a European scale, where large parts are under-investigated, but in a well investigated country like the Netherlands the results are poor – and for butterflies in some cases even wrong– in which case the gap closures have to be removed manually.

Occupancy modeling

Occupancy modeling has been discussed extensively in chapter 4. The basic idea is that a higher observation effort implies a higher probability to detect a species, so variation in observation effort over the years can be directly translated into variation in species detectability. Records from replicate visits to a site allow estimating detection probability separately from the probability of occurrence (Kéry et al. 2010, Van Strien et al. 2010). Examples for the resulting map for *Hipparchia semele* for 1990 and 2010 are shown in figure 4.5.

Even for a well investigated group like butterflies in a well-investigated country like the Netherlands, distribution maps (showing positive records of a species) are still far from perfect. At the same time the need for complete maps, both in time and in space, has increased considerably. Such maps are needed at different levels. The obligations of the Habitats Directive require distribution and range maps at a 10 x 10 km square resolution over six year periods (Evans and Arvela, 2011), which is easily achieved for the three remaining butterfly species listed (of the six species mentioned on the Habitats Directive that once occurred in the Netherlands, *Phengaris arion*, *Euphydryas aurinia* and *Coenonympha hero* being extinct, leaving only *Phengaris teleius*, *P. nausithous* and *Lycaena dispar*), but for some other species groups with a much smaller number of active volunteers (like Mollusca), this can still be hard to achieve.

As butterflies are relatively easy to recognize and their habitat requirements are well-known, an inventory of at least 90% of the sites should be possible in most European countries for the butterflies listed on the Habitats Directive. In many European countries national and local governments support volunteers and, in return, obtain a large amount of high-quality data, allowing them to fulfill all requirements. It is advisable that the remaining countries of the European Union also provide support for the involvement of volunteers in data collection, thereby providing a solid basis for future reporting.

Trend in distribution

Chapter 2 shows the first attempt to find a standardised method to establish a distribution trend for butterflies in the Netherlands. Using a set of reference species (chapter 3) already constituted a considerable improvement over former approaches, and the results were presented in five-year periods by Van Swaay (1995).

Occupancy modelling offers the best results so far with the additional insights into distribution dynamics and trends of colonisation and persistence (chapter 4). As a result, this method has now been used successfully for several species groups in the Netherlands for the reporting of the Conservation Status of species of the annexes of the Habitats Directive (available from mid-2014 on http://bd.eionet.europa.eu/article17/index_html/speciessummary). It is also useful for the compilation of Red Lists.

The results of occupancy modelling could be further improved:

- Delete double records.
  Butterfly records in the Netherlands have been collected in many different ways and thus can enter the National Database Flora and Fauna via several different routes. This leads to a number of double or even triple records: the same observation occurs several times in the database, but each time with a slightly different reference. Evidently, these duplications influence the detection probability and thus the occupancy and its trend.
- Avoid 'me-too' observations.
  Recorders have always wanted to see (and in former times: collect) rare species. For this reason they make targeted trips to known locations with rare species. Oftentimes, only these species are recorded (or collected), thus leading to an unreasonably high detection probability. Promoting the recording of complete species lists could be a partial solution to this problem. In addition, volunteers might be encouraged to survey poorly recorded areas on the basis of maps showing recording intensity, possibly in combination with information on the predicted species-specific habitat suitability.
- Extend the number of high quality non-detections.
  As there are large differences in recording between recorders (and from day to day), leading to large differences in usability for occupancy modelling, it would be much better if data were collected in a more standardized way, e.g. by being sure that all observed species have been recorded. That would be feasible with minor adaptations to the online input platforms.

Future developments in tracking changes in butterfly distribution

Butterfly distribution research typically consists of three phases:

1. The exploration or discovery phase. In this time the species list gets updated almost every year and new expeditions bring in additions regularly. Thomas (2005) shows that the dates of discovery of individual British butterfly species are strongly correlated with their range sizes, with the common widespread species being found first. Many countries in the tropics are still in this phase.
2. The atlas phase. The discovery of new species has become a rare event and more attention gets paid to local, regional and national overviews, e.g. by the publication of atlases with dotmaps. In Europe part of the countries in Southern and Eastern Europe are in this phase at present.
3. The monitoring phase. Additionally trends in distribution and population size become available.

In the Netherlands, we have reached phase 3 and a wealth of detailed data is available. What developments can be expected?

- Although the Netherlands is one of the best investigated countries in the world regarding butterfly distribution, even here there are still gaps, depending on the grid size used. On a 5x5 km grid virtually all terrestrial grid cells are visited at least once (between 2010 and 2012 there were records from 1674 gridcells of 5x5 km, even more than the 1667 terrestrial Dutch grid cells, as some records were from migrating butterflies seen from boats), but on a 1x1 km level 14543 cells (40%) did not have a single record in that period, and on a 250x250 m grid 586575 cells (88%) can be regarded as not-visited and un-studied – and then the quality of the data is not even taken into account, as many are one-record-only visits.It will be almost impossible to fill these gaps: often they are in relatively uninteresting parts of the country (from a butterfly volunteer or nature-lover point of view) or they are inaccessible (e.g. private or a closed nature reserve) and it will be difficult to direct volunteers to visit such gridcells. However, unless probability maps show a high probability for a policy-relevant or Red-listed species, there is not much need to try to stimulate volunteers or professionals to visit such sites.

- On the other hand, there is a great need among managers of naturereserves for detailed information on the distribution of butterflies at their sites (Braunisch et al., 2012), especially where it concerns policy-relevant or Red-listed species. And in such cases the data resolution should be even much finer. The Dutch Subsidy system for Nature and Landscape managment (SNL) even demands a scale of 50x50 m (Van Rosmalen, 2012). With the present data, such a precision cannot be achieved. However, there are several possibilities to fulfill these criteria:
  - Data (at least from nature reserves and other important sites) can be collected by professionals. This method is expensive, but has the advantage that there is a reasonable and controllable certainty that all grid cells get visited. But there is a caveat. As shown in chapter 4, each species always has a detection probability, in the case of *Hipparchia semele* this is 0.58 on a top day in the season (chapter 4). This means that on that top day at least three visits have to be made to exclude the presence of this butterfly with 95% certainty ($(1-0.58)^3=0.03$, so below 0,05). But this detection probability is much lower on other days of the flight season (following the Gaussian curve resulting from the flight period as:
    $$\text{logit}(p_{ijk}) = \alpha_k + \beta_1 * \text{date}_{ij} + \beta_2 * \text{date}_{ij}^2$$
    This means that even more visits are needed to exclude the presence of this butterfly with reasonable (95%) certainty. And as a detection probability of 0.58 is the average for the butterfly species which need to be monitored in the SNL system, this applies to many more species.
  - Collecting such detailed data with volunteers and use professionals as backup. There are some major advantages of working with volunteers (Schmeller et al., 2009), with the price and commitment as the most important ones. The quality of data collected by volunteers is more likely determined by survey design, analytical methodology, and communication skills within the schemes rather than by volunteer involvement per se (Schmeller et al., 2009). The biggest risk when involving volunteers is the uncertainty that all sites get visited frequently enough and with enough coverage, but this can be overcome by subsequent filling of the gaps by professionals, leading to an equal quality and more support (Bos-Groenendijk & Wolterbeek, 2013). Of course this doesn't solve the problem of the number of required visits as with professional data collection.
  - Downscaling from probability maps. At this moment probability maps on a scale of 250x250m are available (Van Swaay et al., 2006).Van Swaay (2013) used a combination of real observations, occupancy maps on a scale of 1x1 km and 250x250m probability maps to produce national maps of each butterfly species for two periods (2002-2005 and 2009-2012) in this fine grid. Although these maps prove very useful to answer methodological questions (e.g.whether the monitoring system for the Dutch

Subsidy system for Nature and Landscape management can produce reliable results), they are still too coarse for application in conservation practice.

  - As most observations are nowadays on a scale of 100x100 m or less (see figure 13.1), occupancy models can be used with this finer grid to produce complete maps. This has been tested for three coastal dune areas in the Netherlands (Wallis de Vries et al., 2013). This gave good results for most of the species and the production of good quality annual maps was possible. However, abundant data are needed, and this method is only applicable in well investigated areas. Furthermore the resulting distribution trends – one of the other outputs – are less sensitive to changes than population trends from butterfly monitoring transects. This is illustrated in figure 13.4 in presenting the distribution trend (on a 1 x 1 km scale, based on the results of occupancy modelling) and the population trend (from the Dutch Butterfly Monitoring Scheme) of *Lasiommata megera*. The population trend started to decline immediately, but only fifteen years later this decline became apparent in the distribution of the species.

- Occupancy modelling based on opportunistic data can deliver good quality distribution trends. However the quality is more or less equal to using the Butterfly Monitoring Data, which also deliver population trends (Van Strien et al., 2013).

It seems reasonable to expect these trends to continue in the near future: more volunteers will collect more detailed distribution and population trend data, but professional coordination and backup will remain necessary, as the demands from policy makers and nature wardens for reliable evidence-based information will grow as well.

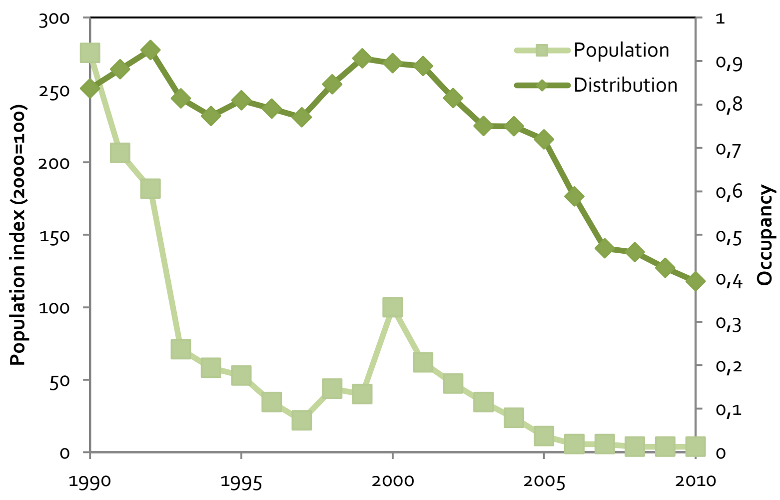


Figure 13.4: Comparison of the Population index (from the Dutch Butterfly Monitoring Scheme) and the distribution (represented by the occupancy) for **Lasiommata megera***.*

Challenges in monitoring butterfly abundance

Already for several decades butterfly monitoring has focused on obtaining quantitative trends of the population size. This has resulted in a great success, with more than 3500 transects in nineteen countries used for the European Grassland Butterfly Indicator in 2012 (Van Swaay et al., 2012).

Although Butterfly Monitoring Schemes are present in a growing number of countries and new ones are being initiated in many places, long time-series are only available for a limited number of countries. The spatial and temporal coverage improves every year, but more development work is needed to achieve complete geographical coverage. This long-term experience with butterfly monitoring in Europe can provide a good template for both other taxonomic groups as well as for other continents, for example in the GEO BON program (Scholes et al., 2008; Pereira et al., 2010).

Table 13.1 provides an overview of the situation in 2012 of the European Butterfly Monitoring Schemes. To be able to draw proper inferences on the temporal population trends at national or regional level, transects should best be selected in a grid, random or stratified random manner (Sutherland, 2006):

- Grid. Locations are placed along a grid over the country. So far, this is only practiced in Switzerland, where all counts are made by professionals.
- Random. Once a recorder registers, a random site in the neighbourhood is provided to them. Random or grid schemes give a more representative sample but often miss rare or threatened species. They are best for recording trends in more widespread species. They are also less practical for involving volunteers and are, therefore, often more costly. Combinations of the two are also possible.
- Free choice. This method is used most frequently in the older schemes (e.g. the UK and the Netherlands). The location of the transect is chosen by the recorder (sometimes together with the co-ordinator), which in some cases has led to the overrepresentation of protected sites in natural areas and the undersampling of the wider countryside and urban areas (Pollard & Yates, 1993), though in Germany Kühn et al. (2008) reported that this effect was not that pronounced. Obviously, in such a case the trends detected may be only representative for the areas sampled, while their extrapolation to national trends may produce biased results. Such bias can, however, be minimized by post-stratification of transects. This implies an *a posteriori* division of transects by e.g. habitat type, protection status and region, where counts per transect are weighted according to their stratum (Chapter 6). Free choice schemes are good for engaging large numbers of volunteers and for covering high quality sites where recorders can see a wide range of butterflies, including rare ones. They are good at detecting site-related trends to inform management on protected sites (e.g. nature reserves).

Grid and random located transects provide the least biased results. However, because of the way they are chosen, the chance that these localities include rare and localised species is small. This means that they don't deliver trends for these rare species, making these Butterfly Monitoring Schemes especially good for biodiversity trends on common and widespread species, but not good for following rare species, often an important part of the focus of butterfly and nature conservation. In general free choice transects are much more focused on rare species and nature reserves, thus also delivering trends on these species.

The number of visits varies from weekly through the main butterfly season in the UK and the Netherlands (26 weeks in theory, the Northwestern European climate leading to an average of 17-19 effective visits) to 3-5 visits annually in France. In the

Netherlands, transects dedicated to rare species need only be visited during the expected flight period of the species.

In normal transects, weekly counts cover the entire flight period of every species and can be used to estimate population trends per transect over time. However, weekly visits may be too demanding for observers. If the only objective is to produce large-scale (e.g. national) trends, the effort may be reduced by having fewer visits (Heliölä & Kuussaari 2005; Roy et al. 2007). Such a reduced-effort scheme is now active in the UK for the Wider Countryside Butterfly Survey, which is based on random 1x1 km squares to detect trends in mainly common butterflies. It is based on only a few annual visits, targeted to the period when most information can be gathered, i.e. three visits in July–August plus in some cases an additional one in May (Roy et al. 2005; 2007). This reduced sampling makes it possible to involve volunteers, but in this case only because of pre-existing networks organised by Butterfly Conservation (UK) and British Trust for Ornithology. In general, many more transects will be needed in a reduced effort scheme than in a traditional scheme.

If transects are selected at random or in a grid, there is a high chance local and rare species will be missed. In the Netherlands we can only calculate trends for **Boloria selene** ***because of targeted, single-species transects.***

Table 13.1: Characteristics of the European Butterfly Monitoring Schemes as submitted by the national coordinators (situation 2012; Van Swaay et al., 2012).

Country	Starting year	Area represented (w=whole country, r=region)	Average transect length	Number of transects per year 2009-2011 (average or range)	Number of counts on a transect per year (average or range)	Counts by (v=volunteers, p=professionals)	Method to choose sites (f=free, c=by co-ordinator, g=grid, r=random	representative for agricultural grassland*	Nature reserves overrepresented*
Andorra	2004	w	1.5	6	20-30	v	f	yes	no
Belgium - Flanders	1991	r	0.8	10	15-20	v	f	no	no
Estonia	2004	w	1.8	11	7	p	c	no	no
Finland	1999	w	3	65-67	ca 11	v ~70%, p ~30%	free for v	yes	no
France	2005	w	1	611-723	4,4 (1-15)	v	half r, half f	yes	no
France - Doubs	2001-2004	r	1	0	10-15	p	c	yes	no
Germany	2005	w	0.5	400	15-20	v	f	yes	yes
Germany - Nordrhein Westfalen	2001	r	1	0	15-20	v	f	no	yes
Germany – Pfalz (Phengaris nausithous only)	1989	r	0.5	50-87	1	p	c	yes	no
Ireland	2007	w	1.5	190	16.3	v	f	yes	no
Jersey	2004-2009	w	1	0	15-25	v	f	yes	no
Lithuania	2009	w	1.3	14	6-9	v	f	no	no
Luxembourg	2010	w	0.34	30	8.2 (3-11)	v ~10%, p ~90%	r	yes	no
Norway	2009	r	1	9-18	3	v ~42%, p ~58%	g	yes	no
Portugal	1998-2006	w	1	0	3-5	v	f	no	no
Romania	starting up								
Russia - Bryansk area	2009	r	1.2	2-14	3-5	v ~90%, p ~10%	f	yes	no
Slovenia	2007	w	1.3	9-14	6.25 - 7.53	v	c	yes	no
Spain - Catalonia	1994	r	1	60-70	30	v	f	yes	no
Sweden	2010	w	0.65	90	4	v	f	yes	no
Switzerland	2003	w	2 x 2.5	90-95	7 (4 alpine region)	p	g	yes	no
Switzerland - Aargau	1998	r	2 x 0.250	101-107	10	p (civil service)	g	yes	no
The Netherlands	1990	w	0,7	430	17 (15-20)	v	f	yes	no
Ukraine – Carpathians and adjacent parts	1990	r	1	158	5 (2-10)	p	f	yes	yes
United Kingdom	1973 (1976)	w	2.7	819-977	19	v	f	yes	yes

**: assessed by experts opinion. In case a monitoring scheme is not representative for agricultural grasslands and/or nature reserves are overrepresented, it means that the resulting trends may be biased towards non-agricultural areas (often nature reserves), where management is focussing on the conservation of biodiversity. Such a scheme probably underestimates the (mostly negative) trend of butterflies in the wider countryside.*

The power of a Butterfly Monitoring Scheme to detect trends depends on many things, the most important ones being (after Van Strien et al., 1997):

- The year-to-year variance: some species, like the Painted Lady (*Vanessa cardui*), show large fluctuations from year to year, where other species, such as the Meadow Brown (*Maniola jurtina*), only show minor changes in abundance from year to year. This means that for some species it is possible to calculate significant trends much sooner than for other species. Furthermore, for species with more than one generation per year, Van Strien et al. (1997) show that the power of the BMS rises when the counts of the first generation are used instead of those of the second generation, as the year-to-year variance of the first generation of most species is considerably lower.
- The number of sampling sites: the more transects there are for a species, the better a trend can be detected.
- The detection period: the longer a scheme is running, the more species trends can be detected.

As a result of the power analysis of the UK Butterfly Monitoring Scheme, 20 transects appears a good minimum to pursue for each species in each stratum that needs to be measured (Van Strien et al., 1997). A stratum can be a country, habitat type, land use or management type, designation category, etc., or combinations of these. For species that are present at more than 50 sites, a further increase in the number of transects hardly improves the power to detect trends (Van Strien et al., 1997). This means that when starting a new country or regional BMS, the focus should be on gaining as many transects as possible. Once the number of transects is over 50, the co-ordinator could focus on other species or start with stratifying the country (e.g. in habitat types or geographical regions) and try to obtain at least 20 transects for each stratum.

For some species there are simply not enough populations to conduct 20 transects. In such cases the coordinator should aim at getting as many populations covered as possible. Where some of these populations occur in remote locations, single-species monitoring can be used, in which only a few counts are made in the peak of the flight period of the species (Van Swaay et al, 2012).

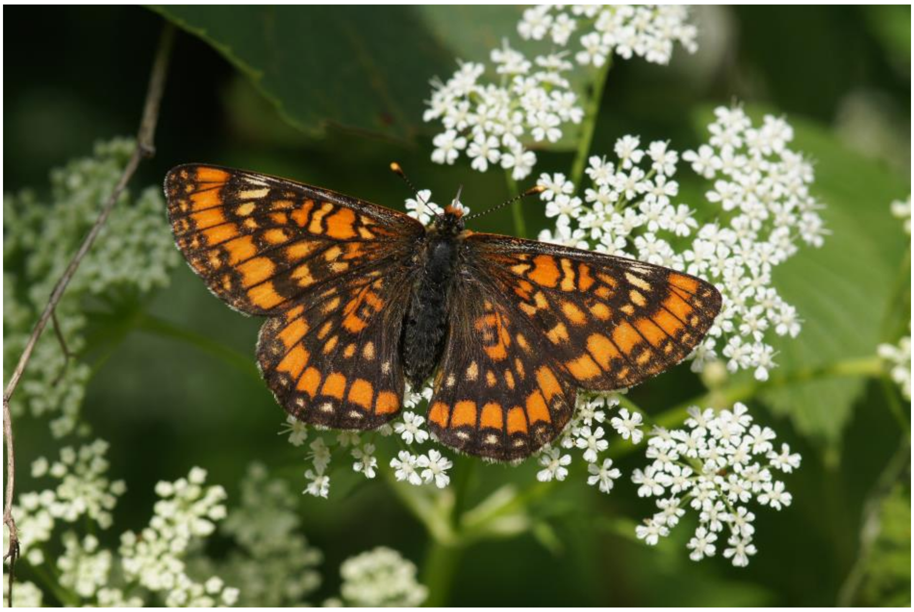
Future developments in tracking changes in butterfly distribution

The first BMS in volunteer-rich countries like the UK and the Netherlands focused on obtaining as many transects as possible. This soon gave good coverage of most species and habitats. However, in other countries with fewer volunteers, it is preferable to focus on a selection of target habitats and species. The following are some options for targeting:

- **Natura 2000 sites**: in the European Union the Natura 2000 network provides a backbone for nature conservation based on a selection of habitats and species mentioned in the annexes of the EU Habitats Directive (see also chapter 11). Many of the important areas for butterflies will be in those Natura 2000 areas, although many other areas will fall outside Natura 2000. By focusing on these areas and the often rare and specialised species in them, most common and widespread species will also be included. The disadvantage is that the resulting trends do not give any information on the situation in the wider countryside, which would be desirable from a policy perspective.
- **High Nature Value Farmland**: it is clear that the highest number of butterflies and species is found on semi-natural grasslands, typically on

High Nature Value Farmland (Opperman et al., 2012). By focusing on these habitats and land-use types, many of the rarer and specialised butterflies will be covered and with them the more widespread and common species.

- **Selected species**: The other way round would be to focus on a selected group of species such as the species listed in the annexes of the Habitats Directive (in the European Union) or Bern Convention (non EU); or the species considered rare and threatened in the European Red List (Van Swaay et al., 2010).



With limited resources, it can be good to focus on a selected group of species, e.g. those of the Habitats Directive, like this* Euphydryas maturna*.

However, in a situation with limited funding and a low number of volunteers, the focus for setting up a new Butterfly Monitoring Schemes should be on:

- Coordinate volunteers: visit local nature conservation groups, use social media, local papers etc. to reach as many volunteers as possible. Using volunteers not only rises the number of transects, they are also important ambassadors of butterflies and their conservation in their local communities, forming a basis for the conservation of butterflies and their habitats.
- Many short transects close to the working or living places of volunteers are better than a few long transects in far-away nature reserves. Even for busy people it is possible to have one or two transects close to their homes and/or working place. Although volunteers tend to want to count in nature reserves with special species, it is the short transects in the urban or agricultural areas that make up the core of the Butterfly Monitoring results for the common and widespread species, allowing research and trends.
- Link up with the international butterfly monitoring community, in Europe via Butterfly Conservation Europe. It is a place to learn from the experience of others. Europe has a wealth of different cultures and ways of tackling the problem of butterfly monitoring with volunteers, so there is a good chance of finding common ground.

Indicators

Indicators are important tools to assess environmental change and the impact of Government policies. They are particularly important to assess progress with the EU Biodiversity Strategy and the goal of halting biodiversity loss by 2020.
Good indicators to measure biodiversity changes should have the following qualities (European Environment Agency, 2007):

1. Policy relevant
2. Biodiversity relevant
3. Measure progress towards target
4. Well-founded methodology
5. Broad acceptance and intelligibility
6. Data routinely collected
7. Cause-effect relationship achievable and quantifiable
8. Spatial coverage, ideally pan-European
9. Show temporal trend
10. Country comparison possible
11. Sensitivity towards change

Butterflies meet most if not all of these criteria and have been selected as a high priority for the development of European indicators under the SEBI 2010 process (European Environment Agency, 2007). Butterfly Conservation Europe has tested the development of a pan-European Butterfly Indicator and has so far produced two indicators: the indicator on European grassland butterflies (chapter 7) and the Climate Change indicator (chapter 8).

The **indicator on European grassland butterflies** was first developed in 2005. It is based on the European trend of 17 grassland butterflies: species that European butterfly experts considered to be characteristic of European grassland and which occurred in a large part of Europe, covered by the majority of the Butterfly Monitoring Schemes and having grasslands as their main habitat (Van Swaay et al., 2006). National population trends from the Butterfly Monitoring Schemes are combined to form supra-national species trends. These trends per butterfly species are then combined into an indicator: a unified measure of biodiversity by averaging indices of species in order to give each species an equal weight in the resulting indicators. When positive and negative changes of indices are in balance, then we would expect their mean to remain stable. If more species decline than increase, the mean should go down and vice versa. Thus, the index mean is considered a measure of biodiversity change.
The most recent update showed that grassland butterflies have declined by almost 50% since 1990 (van Swaay et al., 2012). Because the indicator is constructed from national trends of typical grassland species, it cannot be disaggregated into grassland types. This would be a useful development for the future.
The Grassland Butterfly Index makes a good complement to the Farmland Bird Index (Gregory et al., 2005), because butterflies are far more specialised to grasslands and are more sensitive to changes in the quality of these habitats, which are crucial for biodiversity. They also operate at smaller spatial scales and are thus sensitive to site management. In comparison, farmland birds are better indicators of arable and mixed farms, and large spatial scales.

The **Climate Change Indicator** uses the principle of the Climate Temperature Index (CTI, Devictor et al., 2008). The preference of a species for a specific climate can be expressed by the long term average temperature over its entire range. This is called the Species Temperature Index (STI). The STI was calculated for each

European species, using the European distribution atlas of Kudrna (2002) and the Climatic Risk Atlas of European Butterflies (Settele et al., 2008). The number of butterflies of each species occurring at a certain site in a certain year can be described as a community. As each species has its own specific STI (Species Temperature Index), a Community Temperate Index (CTI) can be calculated as the average of each individual's STI present in the assemblage. A high CTI would thus reflect a large proportion of species with a high STI, i.e. of more high-temperature dwelling species. This way, the CTI can be used to measure local changes in species composition. If climate warming favours species with a high STI, then the CTI should increase locally (Devictor et al., 2008; Devictor et al., 2012). Chapter 8 shows that butterfly communities have shifted northwards by an equivalent of 114 km in 20 years, whereas the temporal trend in temperature has shifted north by 249 km, showing that butterflies are lagging significantly behind climate change (Devictor et al, 2012).

Future developments in indicators
Both indicators could be improved and extended:

- The Grassland Butterfly Indicator could use a wider variety of grassland butterflies, thus improving the quality especially at the northern and southern edges of Europe, where the habitat requirements of species start to change.
- Restrict the Grassland Butterfly Indicator to transects (or parts of transects) on grassland alone to avoid bias from other habitat types.
- Extend the use of indicators to other habitat types. A woodland indicator would be the most logical follow-up, as it includes many species, some of them Europe's most threatened butterflies (e.g. *Coenonympha hero* and *Euphydryas maturna*, both also mentioned in the annexes of the Habitats Directive).
- Developing other environmental change indicators. The Climate Change Indicator is an example of a community indicator. Such indicators can also be developed for other environmental variables, for example the soil nitrogen, acidity and moisture indicators, as extracted from chapter 9. By adding an indicator for the effect of abandonment on butterflies, these indicators could span the most important environmental challenges for both butterflies and humanity in the next decades: climate change, intensification of agriculture (via a Nitrogen indicator) and abandonment. These would be highly valuable tools for European policy makers to monitor the effects of their efforts to preserve biodiversity.

Towards effective butterfly conservation

Effective species conservation as described in chapter 1 relies on a chain of information linking distribution – trend – causes – conservation – communication. Supporting volunteers to collect information on distribution and trend has proven to be a highly effective way to work on the first two pillars. The chapter in parts I and II show that these are supported by a solid scientific basis, providing a reliable source of information.

For conservation to be effective, however, more information is needed on the causes and mechanisms behind the reported changes. There is good and detailed autecological information available for a number of countries, but surprisingly little for the Netherlands (at least compared to a country like the United Kingdom), mostly as a consequence of lack of funding. This leads to the fact that we know very well where the butterflies are and which ones are declining and at what rate, but we lack scientifically sound method to stop this. Much more autecological and conservation research on butterflies is needed to render the investments in the collection of volunteer based distribution and trend data more effective for conservation. Such research should focus on questions like the metapopulation structure of populations of threatened species (making it clear whether it is better to invest in connecting nature reserves or enlarging them), finding the bottlenecks that lead to the decline of species or the effects of environmental pressures on butterflies and their larvae and how to counteract these by taking conservation measures on the ground. Up to that moment the Do's and Don'ts for butterflies of the Habitats Directive of the European Union (Van Swaay et al., 2012) can provide a helpful proxy.
Chapters 8 and 9 reveal basic information on the effects of climate, nitrogen, acidity and moisture on our butterflies. This is a sound basis to explore hypotheses explaining the decline of many species.
On the much coarser European scale the description of their main habitats (chapter 10) in combination with their most important areas for conservation and protection (chapter 11) forms a start for basic butterfly conservation. These Prime Butterfly Areas have been described in greater detail for Bulgaria (Abadjiev and Beshkov, 2007) and Serbia (Jakšic, 2008). Especially in Eastern and Southern Europe, such more detailed descriptions may bring together all expertise on butterflies and identify the main areas to protect them. In many of these countries only a limited number of people and funding is available compared to Northwestern Europe, and Prime Butterfly Areas can help focusing the limited resources.

Butterflies are sensitive indicators of habitat management and measures to stimulate them will also help a wide range of other species, especially insects which form the largest component of biodiversity. For the species of the Habitats Directive the most important management principles can be summarized to (Van Swaay et al (2012) :

1. Manage at a landscape scale.

Butterflies usually occur in a network of local populations between which there is some exchange of adults to form a metapopulation (Hanski, 1999). Progressive loss of habitat suitability across a landscape, or new barriers to dispersal, can lead to loss of local populations and eventually regional extinction of a species through the breakdown of metapopulations. Management should, therefore, aim to maintain such population networks across the landscape, accepting that not every locality may be occupied at any one time (though some core sites should be).

2. Maintain active pastoral systems.
Grassland is the single most important habitat for butterflies and abandonment is the biggest single threat (WallisDeVries & Van Swaay, 2009). Abandonment can temporarily lead to good conditions for many species, but will soon lead to scrub encroachment and eventual loss of suitable breeding conditions as open grassland turns to woodland. The maintenance of open grassland is thus essential, usually by the maintenance of active traditional pastoral systems, including livestock grazing and hay cutting. Socio-economic conditions will need to be considered to ensure that such pastoral systems survive.

3. Manage for heterogeneity.
Grassland butterflies each have their own specific habitat requirements, so management should aim to provide a range of conditions, often based around traditional land use patterns. Some species require short vegetation, while others require taller vegetation. Others still require small-scale mosaics of vegetation types. Managing for spatial heterogeneity across a landscape is thus essential to conserve the full range of typical species (Fahrig et al., 2011).

4. Avoid uniform management (especially in hay meadows).
Butterfly populations can be reduced, or may even become extinct, following intensive and uniform management, notably hay cutting. Cutting dates should be varied as much as possible within Natura 2000 sites so that not all areas are cut within a narrow time window. Ideally, a mosaic of small-scale cutting should be implemented, mimicking traditional management before mechanisation (Konvicka et al., 2008; Cizek et al., 2011).

5. Habitat mosaics are crucial.
Many butterflies use resources found in a range of habitat types and require mosaics of different habitats in the landscape (Marini et al., 2009). For example, some species breed along scrub or wood edges and need a mixture of scrub and grassland (e.g. Turner et al., 2009). Other species may lay eggs in one type of habitat and use nectar resources in another, as *Argynnis aglaja* in the coastal dunes in the Netherlands. The spatial scale of the mosaic will vary from region to region, and will often depend on the traditional land use pattern. Sometimes it will consist of small fields with small blocks of scrub or woodland, while in more extensive landscapes the mosaic may be very large scale.

6. Active woodland management is often essential.
Most woodland butterflies require some form of active management (Freese et al., 2006; Streitberger et al., 2012) and this is essential for the survival of several threatened species. Management can either be regular thinning or rotational coppicing or planting. Some species also require the maintenance of open habitats within woodland, such as sunny clearings or paths/tracks. Traditional management is often a useful guide to suitable management, but may need to be adapted to suit modern timber markets.

7. Monitoring is essential.
Some form of biological monitoring of Natura 2000 sites is essential to ensure management is maintaining the designated features. Butterflies are a sensitive indicator group that can be used to assess change (both positive and negative) and inform decision making. Many butterflies are easy to identify and there are often local volunteer groups or societies that can help provide data. Monitoring can be as simple as successive species inventories, or can be structured around formal sampling procedures such as butterfly transects. The latter are more time consuming but can provide accurate population trends that can identify deleterious changes at an early stage.

From expert judgement to evidence-based conservation

Butterfly conservation has come a long way from 'common sense' (or 'expert judgement'). The number of publications on the topic keeps increasing every year (figure 13.5) and more and more bricks are added to construct an evidence-based form of conservation.

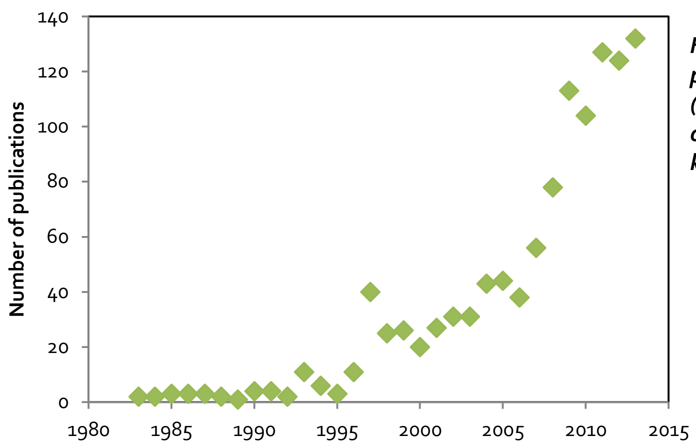


Figure 13.5: Number of scientific publications per year with the words (butterfly OR butterflies AND conservation) in their title, abstract or keywords (source: Scopus.com).

Butterfly monitoring and the indicators based on monitoring data have been shown in this thesis to provide excellent building blocks to track the effects of nature conservation on the main challenges that European butterflies face: climate change, intensification and abandonment. As such, they may become highly valuable for European policy makers to support decision making on the preservation of biodiversity.

However, bringing the message of the ways towards effective conservation to the ground is as yet our biggest challenge. Knowledge is good and important, but dissemination to the wider public as well as to those responsible for nature conservation, is outside the normal scope of many scientists and conservationists. Although not part of this thesis, communication from the results of scientific research as well as the general principles mentioned above, certainly is a vital point in saving butterflies

This thesis illustrated the importance of volunteers and citizen science: without them only small-scale (though detailed) research would be possible and large-scale effects could be hard to prove (or incomparably more expensive). Butterfly conservation cannot move on without their invaluable efforts. Keeping their interest and attention in a rapidly changing world provides new challenges to nature conservation organisations.

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Summary

This thesis consists of three parts: tracking butterfly distribution changes, tracking butterfly abundance changes and how to use this knowledge for their conservation.

The first part discusses several methods to track changes in the distribution of butterflies. Chapter 2 is a follow up of the Dutch Butterfly Atlas (Tax, 1989). It describes a method to follow changes in butterfly distribution up to 1985 by calculating the percentage of the total number of investigated squares where the species was reported in a five-year period. Although this method works well for some species, it does not work well for very rare species. Also the change of method in 1980 – from butterfly collectors to field observations – resulted in errors.

The method presented in chapter 3 using reference species, is a step ahead and was successfully used in the second distribution atlas of Dutch butterflies in 2006 (Bos et al., 2006). The use of occupancy models is a new step forward. Chapter 4 shows the results for the use of this method using distribution data from the Grayling (*Hipparchia semele*) on a 5x5km scale from 1950 onwards (figure i). An interesting added value of these models is the information on survival and colonisation as well as distribution maps with occupancy results per year per square.

As an example figure i shows the trend of *Hipparchia semele* for the three methods in one graph. Occupancy modelling is the superior method, but the use of reference species can be an easy way out when data are lacking or insufficient computer power is available for occupancy modelling.

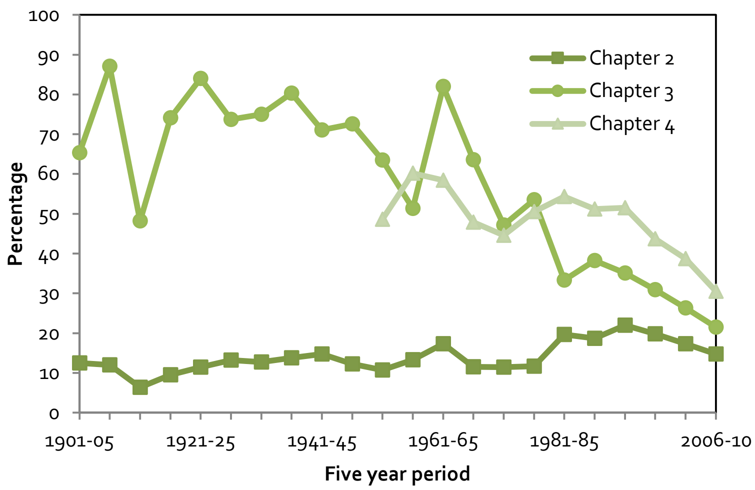


Figure i: Comparison of the trend of Hipparchia semele *following the three methods described in Part I summarised to five-year periods:*

- *Chapter 2: the percentage of the total number of investigated squares where the species was reported.*
- *Chapter 3: corrected with reference species.*
- *Chapter 4: occupancy (only available from 1950 onwards).*

The second part of this thesis focuses on trends in butterfly abundance. Chapter 5 gives a review of butterfly monitoring in Europe and how it can be applied. Two of the main problems before trends can be calculated with the program TRIM are how to arrive at a good estimation of the number of butterflies on a transect in spite of large variartion in monitoring intensity per transect, and how to correct for the fact that butterfly transects are not randomly or irregularly distributed over the country (chapter 6). Combining national butterfly trends to produce a European indicator (chapter 7) is an important step to make changes in butterfly numbers available to policy makers in one single graph (figure ii).

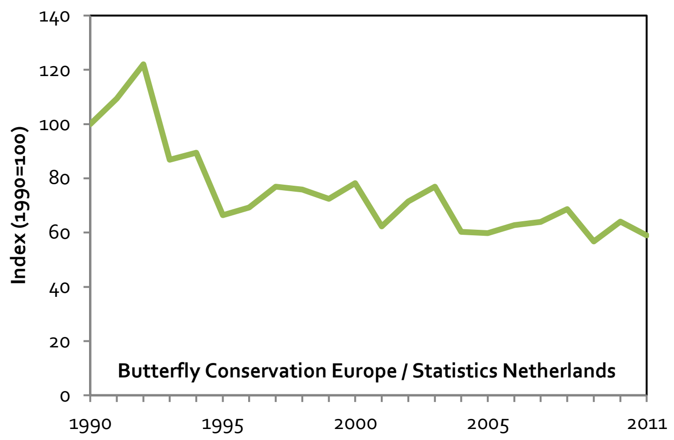


Figure ii: European Grassland Butterfly Indicator (EEA, 2013).

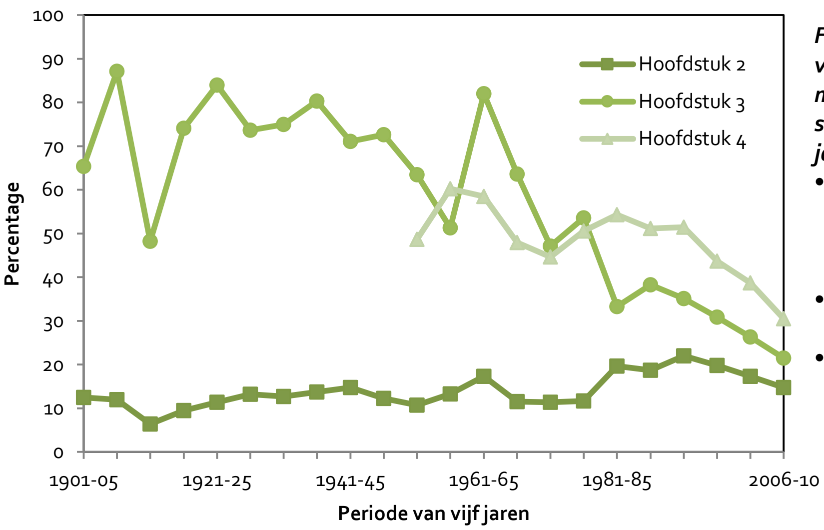
As grasslands are the most important habitat for butterflies, grassland butterflies are the first – and so far only – group for which such an indicator has been produced. Another way to use butterfly monitoring data, which provides an overview of the number of butterflies per site, is presented in chapter 8. Using the average temperature in the distribution area of European butterflies, the weighted changes in butterfly numbers prove to be good indicators for a the reaction of these insect to a changing climate. In twenty years butterfly communities moved 114 km north. Although this may sound impressive, it is by far insufficient to keep up with the speed of the changing climate: to keep the same temperature butterfly communities would have had to move almost 250 km northwards. However butterflies, as short-living insects are still responding much faster than long-living birds, which by now have moved only 37 km within 20 years.

Using data gathered by volunteers and experts from all over Europe – and the Netherlands especially – for the conservation of butterflies is the main item of the third part of the thesis. Chapter 9 describes the relationships between the occurrence of butterflies and the productivity, acidity and moisture of the soil over the vegetation. The results of this chapter can be used in a similar way as the climate change indicator of chapter 8: by using changes in the butterfly community changes in the soil parameters may become apparent. Although rough descriptions of the habitats of butterflies have been available for a long time already, chapter 10 is the first quantification of the preferences of all European butterflies regarding their habitat. To move from a species-based butterfly conservation to an area-based conservation strategy, the production of the European Prime Butterfly Areas is a first step (chapter 11). Finally, chapter 12 discusses the compilation of the latest Red List on European butterflies and what problems arose while developing it. In chapter 13, all previous chapters are discussed in the light of future developments.

Samenvatting

Dit proefschrift bestaat uit drie delen: het volgen van veranderingen in de verspreiding van vlinders, het volgen van veranderingen in de populatiegrootte van vlinders en hoe deze kennis te gebruiken voor hun bescherming.

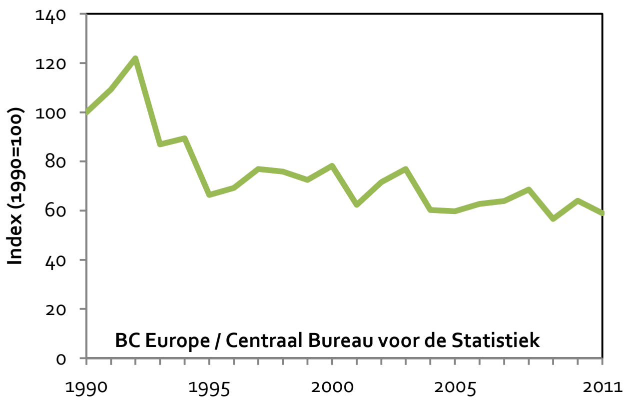
In het eerste deel worden verschillende methoden besproken om veranderingen in de verspreiding van vlinders te volgen. Hoofdstuk 2 is een vervolg op de Atlas van de Nederlands dagvlinders (Tax, 1989). Het beschrijft een methode om veranderingen in de verspreiding van dagvlinders te volgen tot 1985. Hoewel deze methode goed werkt voor sommige soorten, heeft zij moeilijkheden met zeldzame soorten. Ook de verandering van de veldmethode in 1980 – van vlinderverzamelaars naar veldwaarnemingen – leidde tot fouten.
De methode in hoofdstuk 3, het gebruik van referentie-soorten, is een stap vooruit en werd met succes toegepast in de tweede verspreidingsatlas van de Nederlandse vlinders in 2006 (Bos et al., 2006). Het gebruik van occupancy modellen is een nieuwe stap voorwaarts. Hoofdstuk 4 toont de resultaten voor het gebruik van deze methode bij de verspreiding van de heivlinder (*Hipparchia semele*) op een 5x5km schaal vanaf 1950. Een interessant bijproduct van deze modellen zijn de gegevens op de overleving en kolonisatie evenals verspreidingskaarten per jaar per vierkante kilometer.
Als voorbeeld toont figuur iii de trend van de heivlinder (*Hipparchia semele*) voor de drie methoden in een grafiek. Occupancy modellen zijn de beste methode, maar het gebruik van referentie-soorten kan een eenvoudige manierzijn als onvoldoende gegevens of computerkracht beschikbaar zijn voor het gebruik van occupancy modellen.



Figuur iii: Vergelijking van de trend van de heivlinder volgens de drie methoden beschreven in deel I samengevat in perioden van vijf jaren:

- ***Hoofdstuk 2: het percentage van het totale aantal onderzochte hokken waar de soort gerapporteerd.***
- ***Hoofdstuk 3: gecorrigeerd met referentie-soorten.***
- ***Hoofdstuk 4: occupancy modellen (alleen beschikbaar vanaf 1950).***

Het tweede deel van dit proefschrift richt zich op de populatietrends van vlinders. Hoofdstuk 5 geeft een overzicht van Europese vlindermeetnetten en hoe ze gebruikt kunnen worden. Twee van de belangrijkste problemen voordat trends kunnen worden berekend met het programma TRIM, zijn hoe je een goede schatting van het aantal vlinders op een transect komt ondanks grote verschillen in intensiteit van onderzoek per transect, en hoe te corrigeren voor het feit dat vlinder transecten niet willekeurig of regelmatig verdeeld over het hele land zijn (hoofdstuk 6). Het combineren van van nationale vlindertrends naar een Europese indicator (hoofdstuk 7) is een belangrijke stap om veranderingen in vlinder aantallen beschikbaar te maken voor beleidsmakers (figuur iv).



Figuur iv: De Europese Grasland Vlinder Indicator (EEA, 2013).

Graslanden zijn de belangrijkste habitat voor vlinders, en graslandvlinders zijn de eerste - en tot dusver enige - groep waarvoor een dergelijke indicator is geproduceerd. Hoofdstuk 8 laat een andere manier zien waarmee gegevens van de Europese vlindermeetnetten gebruik kunnen worden. Met behulp van de gemiddelde temperatuur in het verspreidingsgebied van de Europese vlindersoorten, blijken de gewogen veranderingen in vlinderaantallen goede indicatoren te zijn voor het aantonen van de invloed van klimaatverandering op deze insecten. In twintig jaar zijn vlindergemeenschappen 114 km naar het noorden opgeschoven. Hoewel dit misschien veel klinkt, is het bij lange na niet genoeg om gelijke tred te houden met de snelheid van het veranderende klimaat: voor dezelfde temperatuur moest je bijna 250 naar het noorden opschuiven. Maar de kort levende vlinders kunnen wel veel sneller volgen dan de langlevende vogels, die pas 37 km zijn opgeschoven in 20 jaar..

Het derde deel van dit proefschrift handelt over het beschermen van vlinders met behulp van gegevens die zijn verzameld door vrijwilligers en deskundigen uit heel Europa - en Nederland in het bijzonder. Hoofdstuk 9 beschrijft de relatie tussen het voorkomen van vlinders en het stikstofgehalte, de zuurgraad en het vochtgehalte van de bodem via de vegetatie. De resultaten van dit hoofdstuk kunnen op dezelfde manier worden gebruikt als de klimaatindicator uit hoofdstuk 8: via veranderingen in de vlindergemeenschap worden veranderingen in de vegetatie en bodemparameters duidelijk. Hoewel ruwe beschrijvingen van de leefgebieden van de vlinders al langer beschikbaar zijn, presenteert hoofdstuk 10 de eerste kwantificering van de voorkeuren van alle Europese vlinders voor hun habitat. Het maakt het mogelijk om habitatspecialiststen te onderscheiden van generalisten. Het overzicht van de belangrijkste vlindergebieden in Europa (Prime Butterfly Areas) is een stap om van soortenbescherming naar gebiedenbescherming te gaan (hoofdstuk 11). Tenslotte wordt in hoofdstuk 12 ingegaan op de resultaten van de meest recente Rode Lijst van Europese vlinders.
In hoofdstuk 13 worden alle voorgaande hoofdstukken besproken in het licht van toekomstige ontwikkelingen.

Curriculum vitae

Chris van Swaay was born on 6 September 1961 in Maastricht. After finishing his HAVO diploma in 1978 at the St Maartencollege in Maastricht, he added two years at the same school to finish his VWO diploma. He studied Biology at the Catholic University (now Radboud University) in Nijmegen with specialisation on plant ecology, animal ecology and aquatic ecology. After finishing his study in 1987 he served eighteen months as civil servant working for Dutch Butterfly Conservation (De Vlinderstichting). After two years teaching animal ecology and working on several projects at Wageningen University, he returned to Dutch Butterfly Conservation in 1990. Here he has worked on many projects, including setting up and coordinating the Dutch Butterfly Monitoring Scheme.
Chris van Swaay is married with Irma Wynhoff, they have two children, Eva and Bosse.
For more information please visit about.me/chrisvanswaay.

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